Research on the Impact of Forest Management in South-West Western Australia

CALM Occasional Paper No. 2/92 JULY 1992

Published by the

Department of Conservation and Land Management
Como, Western Australia

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Published by the
Department of Conservation and Land Management
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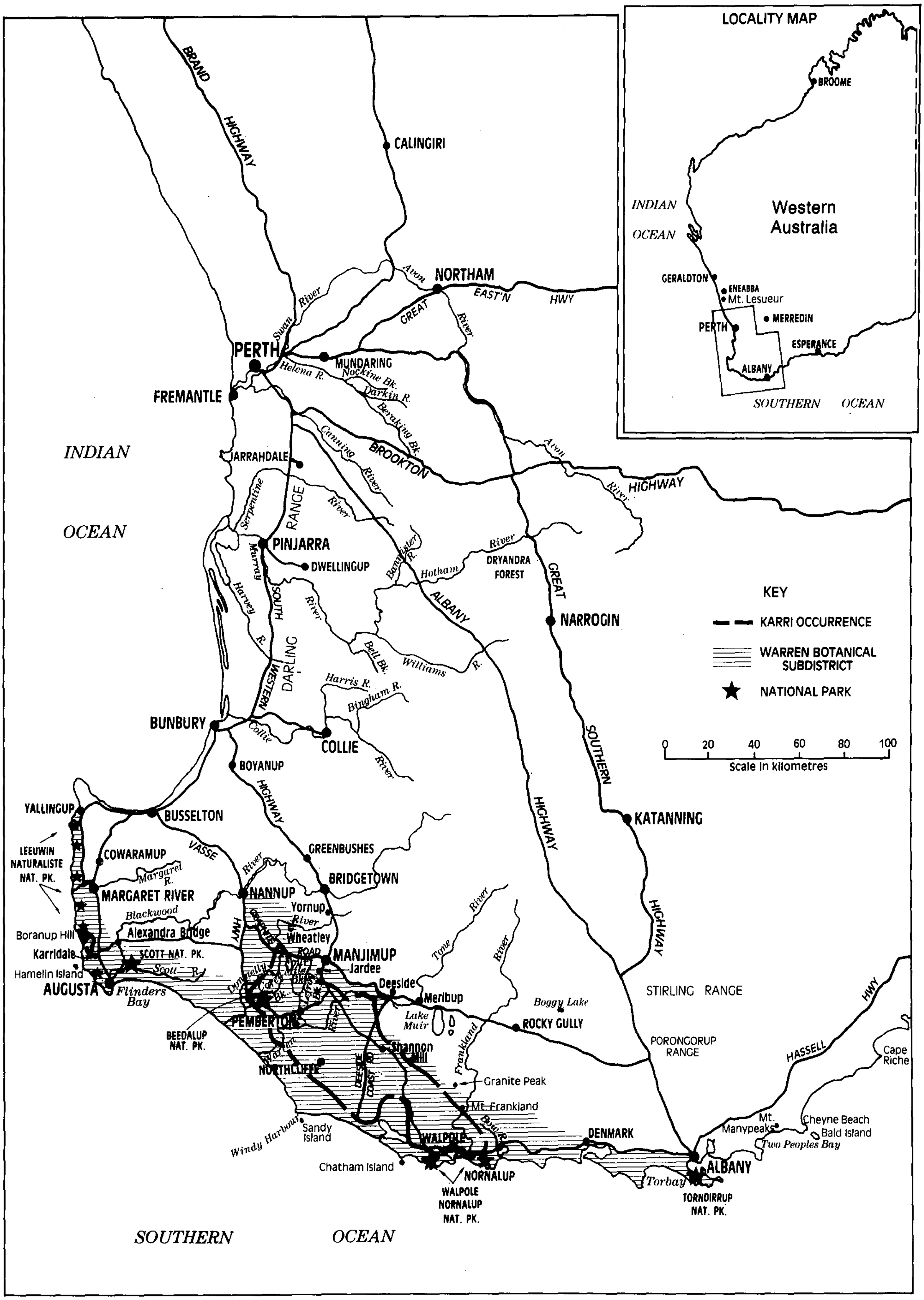
ISSN 1031-4865

Marianne Lewis .... Series Editor
CALM Corporate Relations Division .... Production and distribution



Map of the south-west of W.A. showing karri occurrence and Warren Botanical Subdistrict

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Preface

On 28 September 1988 the Minister for the Environment approved a proposal by W.A. Chip and Pulp Pty Ltd (WACAP) for a renewal of their licence for woodchipping, subject to a number of conditions.

Condition No. 10 states:

> *The proponent shall, in consultation with the Department of Conservation and Land Management, contribute to or participate in monitoring and management with respect to the environmental impact of this proposal to the satisfaction of the Minister of Environment. The detailed objectives and priorities for research into this monitoring and management shall be determined by the Steering Committee for Research on Land Use and Water Supply, following wide consultation, including the public.*

As precursor to agreeing on research requirements with WACAP and the Steering Committee, CALM (the Department of Conservation and Land Management) has prepared a review of existing knowledge written by experts in relevant fields.

The Environmental Protection Authority (EPA) Report and Recommendations[1] on the woodchip industry listed on p. 32 a number of topics that the EPA believed should be addressed. Consequently the following issues have been reviewed:

- Flora of the karri forest and other adjacent vegetation types
- Long term composition of the flora and fauna of the forest in relation to forest management
- The impact of timber harvesting on nutrition of the karri and jarrah forests
- The impact of forest management on the hydrology of the southern forests
- Ecological implications of insect pests in jarrah and karri forests
- Ecological implications of plant diseases in southern forests
- Aquatic fauna of the karri forest

A review of fire is not included here because such a review has been recently published[2].

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[1] EPA (1988). The Western Australian Woodchip Industry. Environmental Protection Authority Bulletin No. 329.

[2] Christensen, P. and Abbott, I. (1989). Impact of fire in the eucalypt forest ecosystem of southern Western Australia: a critical review. *Australian Forestry* **52**, 103-121.

Authors of the reviews were asked to structure their papers around three headings:

- Past research and current knowledge
- Current research
- High priority additional research requirements

As is customary with all CALM scientific publications, these reviews were each refereed by two scientists external to CALM.

Because the authors of the reviews have recommended many additional research requirements, these have been ranked by a group of CALM senior research scientists. Those recommendations considered of highest priority are:

Flora

Production of a Flora of Warren District
Taxonomic studies on poorly known plant groups
Enhanced survey effort, especially surveys of poorly known areas

Effects of timber harvesting and regeneration on flora and fauna

Study effects on selected vertebrates and invertebrates
Study hollow formation in karri, marri and jarrah
Study effects of thinning on fauna

Nutrition

Quantify fertilizer response during early growth of jarrah and karri
Extend later age fertilizer experiments in jarrah and karri to a greater range of sites
Research role of prescribed burning in nitrogen economy of karri forest

Hydrology

None

Insect Pests

Study impact of repeated chronic defoliation on nutrient levels in and physiological condition of jarrah
Clarify reasons for the initiation and continuation of insect outbreaks
Study insect pests of *Eucalyptus globulus*

Plant diseases

Develop a database of susceptibility of plants to *Phytophthora* species and determine long term effects of infection on community diversity
In addition, the Group believed that research on wood rots (not covered in the review) was a high priority

Aquatic fauna

Develop an inventory of aquatic invertebrates
Study impact of logging and regeneration on aquatic communities

I am confident that these reviews provide the essential technical background information necessary to allow agreement on the allocation of research resources.

Andrew A. Burbidge
Director of Research
January 1992

CHAPTER 1

Flora of the Karri Forest and other Communities in the Warren Botanical Subdistrict of Western Australia

S.D. Hopper G.J. Keighery and G. Wardell-Johnson

Abstract

A review is provided of current knowledge, ongoing research and future research requirements for the Warren Botanical Subdistrict flora. Although a thorough survey of the flora has yet to be undertaken, current literature, herbarium collections and unpublished survey data indicate that some 1947 taxa (species, subspecies or varieties) have been collected. This total comprises 1628 native and 319 introduced taxa. Important families include Papilionaceae (165 taxa), Orchidaceae (152), Myrtaceae (125) and Proteaceae (115). The largest genera include *Acacia* (64 taxa), *Stylidium* (55), *Caladenia* (46), *Leucopogon* (41), *Eucalyptus* (26) and *Drosera* (24). The Subdistrict has 99 known endemic taxa, and many more near endemics or geographical outliers. It is the most important centre of endemism for conservative relictual high rainfall taxa in the State.

There are 19 species of Declared Rare Flora, 46 poorly known taxa in urgent need of further survey to assess their conservation status, and 16 adequately surveyed taxa additional to those declared as rare that require monitoring. Coastal heath, granite outcrops, swamps and woodlands contain the majority of endemics and threatened taxa. The main karri forest has few of the endemics, and none of the Declared Rare Flora nor those requiring monitoring.

The most important priority for future research is the production of a published Flora of the Warren Subdistrict. Such a project would stimulate taxonomic and biogeographic research, and enable a better assessment of conservation requirements to be made. Survey and biological studies of threatened taxa and of poorly known and rare habitats at risk are also needed to design appropriate management initiatives.

INTRODUCTION

This paper summarizes existing knowledge of the flora of the Warren Botanical Subdistrict (*sensu* Beard 1980). It presents a list of the known vascular flora, refers to current research projects, and identifies priorities for future research. The work was done primarily as a contribution to a review of research on the impact of forest management in south-west Australia conducted by the Department of Conservation and Land Management, so that it might monitor and manage the woodchip industry with the best available scientific information.

The review of floristic research and plant list may also be of more general interest.

We chose the Warren Botanical Subdistrict as the geographical area of investigation because it includes all karri (*Eucalyptus diversicolor*) forests likely to be harvested for timber, as well as other community types (Table 1). It is also a geographical area used by botanists in specialist literature, and therefore was convenient for data extraction. Examining floristic data for the whole Warren Subdistrict has enabled placement of those components confined to karri forest in a regional perspective.

The Warren Subdistrict extends over 300 km from Yallingup on the Leeuwin-Naturaliste Ridge to Albany on the south coast. It is bounded by the sea to the south and west, while the northern boundary is drawn where *E. diversicolor* ceases to be a significant component (Beard 1980: p. 55). Thus, from Yallingup the northern or inland boundary runs south-south-east to Alexander Bridge on the Blackwood River, then east-south-east across the top of the Scott River Plain to the Donnelly River where it bends due north almost to Nannup, east to Wheatley, south to Jardee, and meanders east-south-east thereafter to Deeside, Granite Peak, Mt Frankland, Denmark and Albany.

The Subdistrict covers 8323 km^2, of which an estimated 31 per cent was cleared for agriculture a decade ago (Beard and Sprenger 1984).

Table 1

BEARD'S (1980) SUMMARY DESCRIPTION OF SALIENT FEATURES OF THE WARREN BOTANICAL SUBDISTRICT

Warren Subdistrict

District name

Geographical, after river of same name (Diels 1906).

Tall forest of karri (*Eucalyptus diversicolor*) on deep loams, forest of jarrah-marri (*E. marginata-E. calophylla*) on the leached sands. Extensive paperbark (*Melaleuca*) and sedge swamps in valleys.

Climate

Moderate 'mediterranean'; winter, precipitation ranges from 650 to 1500 mm per annum, essential feature is short dry season of only 3-4 dry months.

Geology

Archaean granite and infolded metamorphic rocks of the Yilgarn Block.

Topography and soils

Dissected undulating country of small relief, hard setting loamy soils alternating with leached sand soils.

Boundary

The northern boundary is drawn where *E. diversicolor* ceases to be a significant component.

CURRENT KNOWLEDGE

In common with most areas of W.A., neither a comprehensive floristic list nor a full Flora of the Warren Subdistrict has been published. However, scientific knowledge of the composition and distribution of plants in the Subdistrict has accumulated with general botanical exploration of the south-west (see Beard 1981 for a succinct review) and a small number of specialist projects. Information on Warren Subdistrict plants thus appears in more general works on Western Australian vascular flora such as Blackall and Grieve (1954-88), Erickson *et al.* (1973), Beard (1981) and Green (1985). The same applies to lichens (Richardson and Richardson 1982; Sammy 1985), larger fungi (Hilton 1982, 1988) and bryophytes (Wyatt and Stoneburner[1]). Field guides to selected groups of plants provide more detailed information on some species, e.g. banksias (George 1984; Taylor and Hopper 1988), orchids (Hoffman and Brown 1984), and eucalypts (Boland *et al.* 1984; Brooker and Kleinig 1989).

Current botanical information facilitating the identification of some families is becoming available through publication of the *Flora of Australia.* For example, Volume 45 deals with the Liliaceae and Haemodoraceae, Volume 46 with the Iridaceae (George 1986, 1987)

Vegetation and floristics

A few published works deal specifically with Warren Subdistrict vegetation and flora. Beard (1980) provided a brief outline of key features of the Subdistrict (Table 1). Later (Beard 1981), he subdivided the Subdistrict into five vegetation systems. Such systems each constitute a series of plant communities recurring in a sequence or pattern linked to landform, soil and/or geological features. The five in the Warren Subdistrict correspond to the main karri forest (Nornalup System), the more diverse communities to the east where karri is on the highest ground (Denmark System), the low woodlands, heaths and swamps inland from coastal areas (Scott River System), and coastal heaths, peppermint woodlands and low forests between Yallingup and Walpole (Boranup System) and from Walpole to Albany (Torndirrup System). Greater elaboration of these systems and their component communities was provided by Smith (1972-1974) and Beard (1979) in the three 1 : 250 000 vegetation maps and their explanatory memoirs covering the Warren Subdistrict.

Beard and Sprenger (1984) estimated that tall forests (mainly of karri) originally covered 47.8 per cent of the Subdistrict, jarrah forest on poorer soils covered 23.8 per cent, jarrah-banksia low woodlands 8.2 per cent, reed swamps 5 per cent, paperbark low woodlands in swamps 3.7 per cent, peppermint scrub 3.1 per cent, and heath on coastal dunes 2.6 per cent.

Christensen *et al.* (1985) provided a useful summary of major vegetation types and a preliminary floristic list for the southern forests of W.A. Their study area corresponded approximately with Beard's (1980) Warren and Menzies Botanical Subdistricts. Seven major vegetation types were mapped, all occurring in the Warren Subdistrict:

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[1] Prof. R.Wyatt and Dr A. Stoneburner, University of Georgia, USA.

- high open forest of karri or karri and marri, less frequently jarrah, blackbutt and yellow, red and Rate's tingles;
- open forest of jarrah and marri;
- woodlands of casuarinas, banksias and Albany blackbutt;
- low woodlands of banksias, melaleucas, peppermints or eucalypts (on extreme sites);
- closed scrub - heath on swamps or coastal dunes;
- open scrub - heath;
- sedgelands on peaty subcoastal flats.

In addition, these authors singled out granite monadnocks and waterways and wetlands as special sites of unusual and structurally complex vegetation.

For the *c.* 900 plant species of the southern forests listed, the highest numbers were recorded in the open forest and woodland communities, while the least were in the sedgelands, waterways and wetlands, and closed scrub communities. Relatively few local endemics and rare and endangered plants were noted (see also Anon. 1982).

Two regional studies of landforms, soils and vegetation cover part of the Warren Subdistrict. McArthur and Clifton (1975) undertook a study of 34 000 km^2 in the Pemberton area, between the coast to the south, Deeside Coast Road in the east, the Donnelly River and Vasse Highway in the west, and Graphite and Renys Roads in the north. Thirteen broadscale soil associations were defined and mapped. Each was discussed in terms of vegetation structure, floristics, and in current and potential land use.

A more comprehensive study was by Churchward *et al.* (1988), who provided a detailed set of five 1:100 000 scale landform and soil maps covering an area between Windy Harbour eastward to Cheyne (Hassell) Beach 80 km east of Albany, and extending inland to latitude 3430'S (as far north as Rocky Gully). Thirty-five units were mapped, based firstly on general geological features (i.e. units developed on granite or unconsolidated sediments, on siltstones and sandstones, on coastal aeolean and fluvial sediments, and on drainage lines), and then on landform (plateau elements, hills and ridges, swampy terrain, dune systems, and major and minor valleys). Further subdivision into individual units was based on local relief, slope and drainage patterns.

Each of the 35 units was described in the accompanying memoir in terms of its physiography, geology, soil morphology and associated native vegetation. The latter was described structurally and dominants were listed for each major stratum. For example, the Gardner Unit comprises granitic coastal hills and ridges with more than 60 m relief. Areas dominated by granite pavements, domes and tors have vegetation described as

> heath communities of *Agonis marginata, Anthocercis viscosa, Verticordia plumosa* and *Andersonia simplex*; rock surfaces often support *Borya nitida*, orchids, mosses and lichens while ferns grow among the granite boulders .

Smooth gently sloping tracts of sandy podzols between granite outcrops or on lower slopes have

> dense heath (2-3 m high) with a wide range of species including *Hakea elliptica, H. trifurcata, Allocasuarina trichodon, Oxylobium cuneatum, Dryandra formosa, Agonis marginata* and *A. parviceps*; there may be scattered thickets of mallee mainly comprising *Eucalyptus angulosa* and *E. cornuta*. The gullies may have a dense low forest of *Eucalyptus cornuta, E. conferruminata, E. calophylla, Melaleuca baxteri* and *Oxylobium lanceolatum*.

An excellent regional perspective of vegetation types and their association with landforms and soils is thus provided by Churchward *et al.* (1988).

The vegetation and flora of some of the islands and adjacent mainland in the Warren Subdistrict were documented by Gillham (1963) and Abbott (1980a, 1980b; Abbott and Watson 1978). Gillham (1963) showed that islands off Cape Leeuwin with colonies of nesting seabirds had more succulents and annuals in their flora than islands lacking such colonies.

Abbott (1980a, 1980b; Abbott and Watson 1978) provided species lists and/or vegetation maps for Hamelin, Sandy, and Chatham Islands, plus for eight headlands and 21 islands or islets near the large peninsula occupied by Torndirrup National Park south-west of Albany. Most of these studies aimed to provide a baseline to monitor future changes in the vegetation and flora of the islands.

The Torndirrup study established that species richness increases with shelter from waves and salt-spray. Thus, the poorest floras were found on exposed island slopes. Richer communities occurred on sheltered sides of islands where more complex vegetation structure developed. Richer sites again were found on both exposed and sheltered headland areas. Abbott (1980b) explained this trend in terms of exposure to salt, seabird activity and fire regimes.

Flora lists were provided for the Lefroy Brook and Four Mile Brook areas (Dames and Moore 1982), and Smiths Brook Nature Reserve (Griffin 1985). These were baseline studies aimed at assisting planning for a dam site and management of a nature reserve respectively.

A survey of rare and poorly known flora of the Leeuwin-Naturaliste National Park by Hopper and Brown was summarized in the draft management plan for the Park (Frewer *et al.* 1987). The Leeuwin-Naturaliste Ridge was found to be particularly rich in endemic species and others of conservation significance. Some 40 species of interest were identified, with orchids prominent among them. These species were arranged in priority order for management action.

A survey of priority flora of the Walpole-Nornalup National Park provided by Wardell-Johnson and Annels was summarized in the draft management plan for the Park (Smith *et al.* 1990). The Walpole-Nornalup National Park was found to be rich in endemic species and those of conservation significance. Thirty-seven species were listed in priority order for management action and taxonomic work. Seven hundred native vascular plant species have been found in the Park which also includes the major population of red tingle (*Eucalyptus jacksonii*), and important populations of three other large locally endemic eucalypts (*E. guilfoylei, E. brevistylis, E. ficifolia*).

Wetland vegetation has been poorly documented in the Warren Subdistrict, with the exception of Congdon's (1981) study of the fringing vegetation of the Blackwood River Estuary. Congdon mapped the vegetation as three major communities and provided detailed descriptions and transects. The major communities recognized were *Baumea* sedge marsh, *Juncus* rush marsh and *Sarcocornia* marsh. The distribution and zonation of these communities in the above order were correlated with increasing salinity. Seventy-seven species were listed for these communities, 72 native and five introduced. While similar to other estuaries in south-eastern Australia, communities on the Blackwood River Estuary were less productive and showed more subtle zonation owing to a lower tidal amplitude.

Evolutionary studies

Churchill (1961, 1968) provided palaeontological and palynological data derived in part from sites in the Warren Subdistrict (Weld Swamp near Shannon Mill, Flinders Bay Swamp at Augusta, Scott River Swamp east of Augusta, and Boggy Lake in Walpole-Nornalup National Park). He documented changes in the abundance of karri, marri and jarrah pollen over the past 6000 years, with periods favouring karri from 4000-3000 B.C., 500 B.C. - 700 A.D., and 1500 A.D. to the present.

Evolutionary and genetic studies of the living flora have been few, but include James' (1979) chromosome number survey in *Stylidium*, a cladistic study of south-western monocalypt *Eucalyptus* by Ladiges *et al.* (1987), and Coates and Sokolowski's (1989) survey of allozyme variation in karri. These works suggest that the Warren Subdistrict contains many relictual taxa (e.g. Rate's tingle *Eucalyptus brevistylis*) whose relationships may be with eastern States congeners as much as with Western Australian relatives in adjacent botanical districts. A similar situation appears to hold in some *Banksia* species such as *B. seminuda* ssp. *remanens* and *B. occidentalis* ssp. *formosa* (Hopper 1989). The evolutionary conservatism and relictual nature of most of the high rainfall zone flora was emphasized by Hopper (1979).

However, some groups do have their centres of diversity and appear to have undergone active recent speciation in the Subdistrict. For example, Burbidge (1984) found a centre of species richness in the Warren Subdistrict in triggerplants (*Stylidium*), with some groups concentrated there (e.g. the leafy-stemmed triggerplants and the true annuals). Rye (1980) showed that *Agonis* and *Hypocalymma* in the Myrtaceae similarly were concentrated in the Subdistrict.

Keighery (1984) found that most groups of wetland monocotyledons, including genera of Cyperaceae, Xyridaceae, Juncaginaceae, Restionaceae and Orchidaceae, were species-rich in the Warren. Recent taxonomic studies in the Orchidaceae (Bates 1984; Hopper and Brown, in press) have accentuated this trend, as many new taxa endemic to the Subdistrict have been discovered.

Thus, it would appear that for woody genera, active speciation has been concentrated in the transitional rainfall zone, while the wetter Warren Subdistrict has provided a refugium for the persistence of relictual taxa. For annuals and perennial herbs, on the other hand, the Subdistrict has been a major centre of speciation.

Community studies

Seven site-based community studies in the Warren Subdistrict have been published or are in press (others are in progress). Two of these were in forest dominated structurally by jarrah, three by karri, and two in coastal heathland.

George *et al.* (1979) included three coastal sites in the Warren Subdistrict (at Torndirrup, Walpole-Nornalup and Scott National Parks) in a comparison of 25 heathland communities throughout south-western Australia. The Warren Subdistrict sites were of average species richness (50-56 per 100 m^2), with less species than lateritic heathland sites such as at Eneabba-Mt Lesueur, Mt Manypeaks and the Stirling Range (77-92 species), but more species than heathlands in the central western wheatbelt (40-44 species). The Warren sites were atypical of the south-western flora as a whole in their taxonomic composition, being relatively low in Myrtaceae and Proteaceae, but rich in Papilionaceae, Goodeniaceae, Epacridaceae and Stylidiaceae.

Enright (1978) studied a small area of coastal heath near The Gap in Torndirrup National Park. He was interested in testing whether podzolization (formation of acidic non-calcareous soils) was explicable owing to the activity of water-soluble iron-complexing leaf compounds. Do certain plant species aid in development of the podzol soil profile, or do podzols select the kinds of plants that can germinate and grow successfully on them?

Approximately equal numbers of podzol and calcareous sites were chosen for the placement of 24 quadrats 3-m^2 to sample the flora. Enright found that podzol sites were characterized by a high frequency of *Andersonia simplex, Lysinema ciliatum, Leucopogon reflexus* and *Dasypogon bromeliaefolius*. Non-podzol (calcareous) sites supported larger populations of *Bossiaea rufa, Olearia axillaris, Pimelea rosea* and *Leucopogon revolutus*. Leaf extracts of common species from podzol, calcareous and transitional sites were tested for their ability to form a complex with iron. Only podzol and transitional species proved to have this ability. Enright concluded that soil chemistry was the major determinant of which species will grow in the early stages of calcareous soil development, but that plants themselves play an increasing role in determining community composition as they remove iron and aluminium and acidify the soil to a podzol.

Strelein (1988) presented an ordination using over 400 sample sites and 100 indicator species in the southern jarrah forest. He defined seventeen site types from this work using the methods of Havel (1968, 1975) and discussed the regeneration, dieback susceptability and productivity of each. Strelein (1988) defined the species to be used prior to analysis and presented a list of 211 species recorded. He suggested that all but one site type (type Q - high quality forest on fertile, well drained loams) have some susceptability to dieback disease.

Inions *et al.* (1990) derived a floristic classification of regenerating karri forest in the Nornalup System of the Warren Subdistrict. They used 204 permanent inventory plots (Campbell *et al.* 1985) and 105 species were sampled. Annuals, herbs and outliers of forest including karri as a component (e.g. Boranup, Manypeaks and Porongurups) were not included in the classification. All sites were in regrowth karri forest. Thirteen community types were defined by cluster analysis, ordination and discriminant analysis of the 312-m^2 quadrats. Inions *et al.* (1990) found that community types varied substantially in productivity as measured by age-standardized top height and this in turn was related to climatic and edaphic factors. Variables relating to rainfall distribution, radiation levels, soil acidity and phophorous levels were found to be the most discriminatory between community types and each differed in stand productivity and in climatic and edaphic variables. The distribution of the community types defined is broadly geographically based (Wardell-Johnson and Christensen, this volume), although overlap occurs within a single landform/soils unit (as defined by Churchward *et al.* 1988).

The schemes developed by Inions *et al.* (1990) and Strelein (1988) each provide a different means of defining a community or site type. Strelein did not use permanent quadrats. Although each of these studies was in jarrah forest or karri forest, overlap is likely between the studies.

Current emphasis in management is on the ecotone between these two forest types (Bradshaw[2] personal communication 1988). Two studies (Bridgewater 1981; Wardell-Johnson *et al.* 1989) have examined ecotones in addition to pure forest stands in the Warren Subdistrict.

Bridgewater (1981) used the Zurich-Montpellier system of vegetation description and classification to define the karri forest boundary near Pemberton. A complete species list (40 species) for the time of the visit is presented for Bridgewater's 26 quadrats.

Wardell-Johnson *et al.* (1989) developed a floristic classification of the Walpole/Nornalup National Park based on 219 quadrats and 233 species. Twelve community types were derived with clustering and ordination techniques and were associated with the landform soils units of Churchward *et al.* (1988). A vegetation map was published which recognized this association (Smith *et al.* 1990). Forest in which karri is a component is separated from other community types at the three-group level in cluster analysis and is the most species-poor of the community types.

Both Inions *et al.* (1990) and Wardell-Johnson *et al.* (1989) provided a means of allocating independent sites to the classification using discriminant functions on species defined as indicators in the analyses (72 and 52 species respectively). Thus sites in one classification can be defined according to another.

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[2] F.J. Bradshaw, CALM, Manjimup.

Thus Wardell-Johnson (own data) found that thirty sites in the Walpole-Nornalup National Park defined as community type 8 (*Eucalyptus diversicolor* forest community) separated into one of two community types (Stoate and Wallace) using the Inions *et al.* (1990) scheme.

Classifications developed in both studies have used similar methods and both schemes can be mapped. All studies have used a similar quadrat size (either 312 m^2 or 400 m^2) but not all studies have used the same components of the flora in deriving the classification or ordination. Thus, although an integration of site-based work in the Warren Subdistrict is desirable, considerable site revisiting will be required.

Burbidge and Boscacci (unpublished) suggested a quadrat size in excess of 400 m^2 to sample 95 per cent of the expected species in a study near Northcliffe. These workers provided species area curves, lists of species and a classification of the 13 quadrats.

Standardization of plot sizes, permanent marking of quadrats and a complete enumeration of the flora in a given quadrat are recommended in further site-based studies in the Warren Subdistrict.

LIST OF THE VASCULAR FLORA

A major aim of this paper was to compile a list of the known vascular flora of the Warren Subdistrict. The list (Appendix 1) is largely the work of GJK, with additional taxa added by CALM Manjimup research staff, Flora Conservation Research Program staff, Dr N.G. Marchant[3] and Dr N. Gibson[4] (personal communication).

The list was compiled by examining all specimens incorporated into collections of the Western Australian Herbarium (up to 1985), Kings Park Herbarium (1985) and CALM's Manjimup Research Herbarium (1988), plus a search of recent Australian taxonomic literature (1950 - 1988) and the compilation of lists in published and unpublished works of ourselves and other Western Australian botanists. A.P. Brown and SDH prepared the orchid list. Records for which no voucher specimen could be traced were rejected.

A total of 1947 taxa is known for the 8323 km^2 of the Subdistrict (Table 2). This includes 1628 native and 319 introduced taxa. This compares favourably with the larger Perth Region (of 10 500 km^2, Marchant *et al.* 1987), which has 2057 taxa (1510 native and 547 introduced).

Families with the largest number of known taxa in the Warren Subdistrict include:

Monocotyledons

Orchidaceae	152
Poacae	100
Cyperaceae	75

Dicotyledons

Papilionaceae	165
Myrtaceae	125
Proteaceae	115
Asteraceae	105
Mimosaceae	65
Epacridaceae	78
Stylidiaceae	60

Table 2

NUMBERS OF NATIVE AND INTRODUCED VASCULAR TAXA FOR THE WARREN BOTANICAL SUBDISTRICT AND THE PERTH REGION (MARCHANT *ET AL.* 1987)

	Native	*Introduced*	*Total*
Warren Subdistrict			
Ferns	17	2	9
Gymnosperm	3	2	5
Monocotyledons	481	99	580
Dicotyledons	1127	216	1343
Totals	1628	319	1947
Perth region flora			
Ferns	23	2	25
Gymnosperms	5	2	7
Monocotyledons	462	191	653
Dicotyledons	1020	352	1372
Totals	1510	547	2057

Especially noteworthy are the unusually large numbers of Orchidaceae, Stylidiaceae and Epacridaceae compared with the State's flora as a whole (Green 1985).

The largest genera include *Acacia* (64 taxa), *Stylidium* (55), *Caladenia* (46), *Leucopogon* (41), *Eucalyptus* (26) and *Drosera* (24).

Endemics and geographical outliers

The known Warren Subdistrict flora has 99 endemics (compared with 43 in the Perth Region). This number is probably conservative, as several groups require taxonomic revision, and groups such as orchids that

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[3] N.G. Marchant, W.A. Herbarium, South Perth.

[4] N. Gibson, CALM, Woodvale.

have been worked on recently have many endemics among the undescribed taxa recognized.

Most of the known endemics are confined to swamplands, coastal heaths and granitic outcrops, with relatively few found in forests. The majority are found in the wettest country between Denmark and Northcliffe, but some are concentrated on the Leeuwin-Naturaliste ridge (9 taxa) and two are known endemics of the Scott Coastal Plain.

Many more species, including karri itself, are near-endemics of the Subdistrict. That is, they are confined to high rainfall areas or moist habitats, and extend outside the Subdistrict in small areas of favourably wet conditions (e.g. Porongurups, Mt Manypeaks for karri; Cape Riche and Yallingup for *Cephalotus follicularis*; swamps at the base of the Darling Scarp for *Actinodium cunninghamii* and *Reedia spathacea*). There are at least 52 dicotyledons in this category alone. Hence, the Warren Subdistrict is a major centre of endemism for wet country taxa in the State.

Another feature of the flora is the large number of taxa that reach the end of their geographic range in the Warren Subdistrict. For example, 56 south coast heathland species reach their western limits between Albany and Denmark (e.g *Eucalyptus angulosa, Calectasia grandiflora, Stylidium hirsutum*). Nine species of the Swan Coastal Plain extend to or have outliers at Cape Leeuwin and the Scott River area (e.g. *Trachymene caerulea, Conostylis candicans*).

Declared Rare Flora and other plants of special conservation significance

The most recent schedule of Declared Rare Flora (*Government Gazette* of 17 May 1991) listed the following 19 Warren Subdistrict taxa (main habitat is given to the right):

Adenanthos cunninghamii	coastal dunes
Asplenium obtusatum	island granite outcrops
Baeckea arbuscula	swamps
Banksia goodii	jarrah low forest
Banksia verticillata	coastal granite outcrops
Caladenia excelsa	Banksia low woodlands
Caladenia harringtoniae	swamps
Caladenia huegelii	jarrah forest
Caladenia viridescens	marri-jarrah forest
Darwinia 'ferricola'	lateritic heath
Diuris drummondii	swamps
Drakaea micrantha	jarrah forest
Grevillea cirsiifolia	jarrah forest
Kennedia glabrata	forest granite outcrops
Kennedia macrophylla	granite outcrops, karri low forest
Isopogon uncinatus	coastal heath
Lambertia orbifolia	jarrah forest
Laxmannia jamesii	swamps
Microtis globula	swamps

Six of these taxa occur in jarrah forest, five in swamps, four on granite outcrops, two on coastal dunes, and one each in *Banksia* low woodlands and lateritic heath. None are known from the main karri forest. A population of *Kennedia macrophylla* atypically occurs in stunted coastal karri (*Apium prostratum* ssp. *'phillipii'* occurs beneath karri in the Porongurups outside the Warren Subdistrict).

An additional eight taxa have been listed on previous schedules of Declared Rare Flora but were found to be more abundant than previously thought. Populations are monitored by CALM staff. Eight other taxa are similarly monitored because they are adequately surveyed and not considered endangered or in need of special protection but could be if present circumstances change. These sixteen monitored taxa and their main habitats are:

Adenanthos detmoldii	swamps
Banksia meisneri var. *ascendens*	swamps
Banksia seminuda ssp. *remanens*	coastal granite
Caladenia arrecta	jarrah forest
Caladenia plicata	jarrah forest
Caladenia interjacens	coastal heath
Caladenia nivalis	coastal granite
Caladenia speciosa	jarrah woodland
Calothamnus graniticus ssp. *graniticus*	coastal granite
Chamaexeros sp.	coastal woodland
Eucalyptus calcicola	coastal dunes
Grevillea ripicola	granite outcrops
Microtis pulchella	swamps
Pentapeltis sylvatica	jarrah forest
Prasophyllum triangulare	jarrah forest
Restio ustulatus	swamp heath

There are 46 poorly known taxa collected from the Warren Subdistrict whose conservation status is uncertain and needs urgent investigation. These include:

1. Taxa presumed extinct (not collected or reliably observed over the past 50 years)

Meziella trifida	Albany
Scaevola attenuata	Albany, ?Cape Naturaliste
Tetratheca elliptica	Bow River

2. Taxa known only from one or a few localities on lands under threat

Actinotus 'laxa'	Walpole
Alexgeorgea ganopoda	Bow River, Mt Frankland
Andersonia auriculata	Quarram, W Denmark, Bow River
Aotus carinata	Scott River Plain
Hemiandra podalyrina	E Northcliffe

Hybanthus volubilis	Margaret River
Isopogon uncinatus	Mt Willyung, Torndirrup
Jacksonia mollissima	Bunbury-Margaret River
Leptomeria ericoides	Mt Willyung, Cowaramup
Leucopogon alternifolius	Scott River, Albany
Leucopogon polystachyus	Manjimup, Nornalup
Restio gracilior	Scott River, Busselton
Schoenus acuminatus	Albany
Schoenus efoliatus	Albany
Schoenus multiglumis	Albany
Sollya drummondii	Denmark
Thysanotus formosus	Nannup
Thysanotus isantherus	Albany

3. Taxa known from one or a few localities on land not under immediate threat

Amperea volubilis	Walpole, Albany
Banksia occidentalis ssp. *formosa*	Black Pt, Torbay
Caladenia evanescens	Walpole, Peaceful Bay
Caladenia huegelii ssp. *redacta*	W Northcliffe
Caladenia winfieldii	Tone River
Boronia crassipes	Albany
Hemiandra glabrescens	Albany, Scott River Plain
Leucopogon bracteolaris	Stirling Range, Albany
Leucopogon multiflorus	Albany
Lomandra ordii	Northcliffe, Walpole
Lysinema lasianthum	Porongurups, Albany
Pithocarpa melanostigma	Albany, Millbrook
Stylidium barleei	Busselton area
Tripterococcus sp. nov.	Scott River, Walpole,

4. Taxa known from several localities, some of which are on lands not under immediate threat

Boronia virgata	Walpole, Denmark
Drosera omissa	Augusta, Busselton
Gastrolobium brownii	Walpole, Albany
Melaleuca basicephala	Scott River
Pultenaea pinifolia	Busselton, Karridale
Restio ustulatus	Busselton, Scott River
Thomasia discolor	Albany area
Thomasia solanacea	Albany, Two Peoples Bay
Villarsia lasiosperma	Busselton, Esperance

A more thorough biogeographical analysis of the present flora list may well highlight other poorly known taxa whose conservation status requires attention. This applies also to the nonvascular flora, which we have not investigated (we do know, however, of at least one moss, *Rhacocarpus webbianus* (C. Muell.) Par., which is only known from Mt Chudalup and possibly Two Peoples Bay - Wyatt and Stoneburner, personal communications.

Weeds

Although less taxa of weeds occur in the Subdistrict than in the Perth Region (Table 2), those present pose major problems to conservation managers. For example, granite outcrop communities rich in endemic species have been successfully invaded following disturbances such as grazing. This is evident in the karri and granite communities of the Porongurups which have been invaded by *Plantago lanceolata, Cirsium vulgare, Briza* spp. and *Trifolium* spp.

In the main karri forest, *Acaena* spp. and *Trifolium* spp. appear to be persistent invaders following disturbance. *Rubus* spp. and *Solanum* spp. are major problems along creeklines.

CURRENT STUDIES

Within CALM, ongoing floristic studies of the Warren Subdistrict are included in projects within the Biogeography, Fire, Flora Conservation and Flora Collections Research Programs.

Christensen is examining long-term fire effects in karri forest near Manjimup. Burrows has established five sets of permanently located quadrats to examine the long-term effects of fire in the Southern Forest Region. Four of these study sites are in the Warren Subdistrict. A list of flora and a set of indicator species for one of these study sites in open jarrah forest has been derived from earlier studies.

Wardell-Johnson is obtaining a complete list of flora from permanently marked quadrats in four study sites in karri forest at Gray forest block. Wardell-Johnson has listed vascular flora from 95 x 400 m^2 quadrats and aims to derive an age series of floristics in community type Shea of Inions *et al.* (1990).

Wardell-Johnson has collected vascular flora from 211 permanently located 600-m^2 quadrats in the Walpole area. This study aims to examine floristic, edaphic and climatic attributes associated with the distribution of four locally endemic forest eucalypts. This study commenced in April 1989.

Gibson is conducting a survey of regional floristic variation in heath and peppermint low woodland communities between Cape Naturaliste and Albany. This work aims to provide a regional context within which proposals for mineral sand exploration and mining may be assessed.

Keighery is compiling checklists of the vascular flora of south coast reserves, and has manuscript lists for West Cape Howe, Torndirrup and William Bay National Parks in the Subdistrict. Other areas are studied opportunistically, with special emphasis on weeds.

Opportunistic surveys of rare and poorly known flora are continuing throughout the Subdistrict by Flora Conservation Research Program staff and regional operations staff. Specialist surveys of orchids, eucalypts and granite outcrop flora are part of Hopper's current research. Many new taxa of *Caladenia*, including 11 endemic to the Subdistrict, are described by Hopper and Brown (in press).

Coates is investigating allozyme variation and mating systems in karri itself with a view to improving the conservation and management of genetic resources.

Ad hoc collections of Warren Subdistrict flora are made by Herbarium staff in the course of taxonomic research. Macfarlane is currently describing the new *Chamaexeros* sp. from Walpole-Nornalup National Park.

N. Malajczuk of CSIRO is studying the biology and systematics of soil fungi of the karri forest.

Professor R. Wyatt and Dr A. Stoneburner of the University of Georgia U.S.A. are compiling a checklist of the bryophytes of the State, and have collected at several sites in the Warren Subdistrict.

HIGH PRIORITY ADDITIONAL RESEARCH REQUIREMENTS

A major requirement is the production of a Flora of the Warren Botanical Subdistrict. This would increase the level of botanical survey and research, and enable clearer definition of taxa in need of management for their conservation.

Concurrent with the production of a Flora is the need for a systematic and standardized site-based survey of the vegetation communities of the Subdistrict.

In view of the large number of local endemics in groups subject to recent taxonomic research (e.g. orchids), it is considered a priority to stimulate taxonomic studies on poorly known groups such as Poaceae, Tremandraceae, Cyperaceae, Epacridaceae (especially *Andersonia* and *Leucopogon*), and Stylidiaceae.

The identification of taxa most in need of conservation initiatives is a priority, and will come only with an enhanced survey effort and biological research program. The latter may enable the categorization of vulnerable species according to life-history attributes. For example, obligate seed regenerators may be the most vulnerable group to fire.

Those areas most in need of survey include the Mt Lindesay granite monadnocks, forest north of Nornalup, the Scott Coastal Plain, coastal heaths between Walpole and Augusta, and granite outcrops throughout the Subdistrict.

Taxa most in need of biological research include Declared Rare Flora and restricted endemics susceptible to dieback, inappropriate fire regimes (e.g. fires at a greater frequency than the time needed for obligate seeders to replenish seed stores), grazing, weed invasion and other disturbances.

ACKNOWLEDGMENTS

We are grateful to colleagues too numerous to mention for assistance in field work, to the Curator of the Western Australian Herbarium for providing access to the collection, to Andrew Brown for additions to the list of orchids, Dr N. Gibson for other additions to the flora list and to Dr N. Marchant and Dr B. Briggs for useful comments on the manuscript. Dr Marchant also thoroughly checked, corrected and added to the species list, a contribution much appreciated.

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Appendix I

List of Flora of the Warren Botanical Subdistrict as known at June 1991

Nomenclature and general systematic arrangement follow Green (1985)
* = Introduced; (E) = endemic to the Warren Botanical Subdistrict

FERNS

LYCOPODIACEAE
Lycopodium serpentinum Kunze
Phylloglossum drummondii Kunze

SELAGINELLACEAE
Selaginella gracillima (Kunze) Alston

ISOETACEAE
Isoetes australis S. Williams
Isoetes drummondii A. Braun

OPHIOGLOSSACEAE
Ophioglossum lusitanicum L. ssp. *coriaceum* (A. Cunn.) Clausen

SCHIZAEACEAE
Schizaea fistulosa Labill.

ADIANTACEAE
Adiantum aethiopicum L.
Anogramma leptophylla (L.) Link.
Cheilanthes austrotenuifolia Quirk et Chambers

PTERIDACEAE
Pteris vittata L.

CYATHEACEAE
* *Sphaeropteris cooperi* (Hook. ex F. Muell.) Domin

DENNSTAEDTIACEAE
* *Hypolepis rugulosa* (Labill.) J. Smith
Pteridium esculentum (G. Forster) Cockayne

LINDSAEACEAE
Lindsaea linearis Sw.

ASPLENIACEAE
Asplenium aethiopicum (Burm.f.) Bech.
Asplenium flabellifolium Cav.
Asplenium obtusatum G. Forster

MARSILIACEAE
Pilularia novae-hollandiae A. Braun

GYMNOSPERMS

ZAMIACEAE
Macrozamia riedlei (Fischer ex Gaudich.) C.A. Gardner

PODOCARPACEAE
Podocarpus drouynianus F. Muell.

PINACEAE
* *Pinus pinaster* Aiton
* *Pinus radiata* D. Don

CUPRESSACEAE
Actinostrobus pyramidalis Miq.

MONOCOTYLEDONS

TYPHACEAE
* *Typha orientalis* C. Presl

RUPPIACEAE
Ruppia megacarpa Mason
Ruppia polycarpa Mason

POTAMOGETONACEAE
Potamogeton ochreatus Raoul
Potamogeton drummondii Benth.

ZANICHELLIACEAE
Lepilaena bilocularis Kirk
Lepilaena cylindrocarpa (Koern. ex Walp.) Benth.
Lepilaena preissii (Lehm.) F. Muell.

POSIDONIACEAE
Posidonia australis J.D. Hook.
Posidonia denhartogii Kuo et Cambridge
Posidonia kirkmanii Kuo et Cambridge
Posidonia ostenfeldii Hartog
Posidonia robertsoniae Kuo et Cambridge
Posidonia sinuosa Cambridge et Kuo

CYMODOCEACEAE
Amphibolus antarctica (Labill.) Sonder
Amphibolus griffithsii (Black) Hartog
Thalassodendron pachyrhizum Hartog

ZOSTERACEAE
Heterozostera tasmanica (Martens ex Asch.) Hartog

NAJADACEAE
Najas marina L.

JUNCAGINACEAE
Triglochin calcitrapa Hook.
Triglochin centrocarpa Hook.
Triglochin minutissima F. Muell.
Triglochin procera R. Br.
Triglochin striata Ruiz. et Pav.
Triglochin trichophora

HYDROCHARITACEAE
Halophila ovalis (R. Br.) J.D. Hook.
Ottelia ovalifolia (R. Br.) Rich.
* *Vallisneria spiralis* L.

POACEAE
Agrostis avenaceae J. Gmelin
Agrostis drummondiana (Steud.) Vickery
* *Agrostis gigantea* Roth.
* *Agrostis stolonifera* L.
Agrostis venusta Trin.
* *Aira caryophyllea* L.
* *Aira cupaniana* Guss.
* *Alopecurus myosuroides* Hudson
* *Ammophila arenaria* (L.) Link
Amphipogon amphipogonoides (Steud.) Vick.
Amphipogon debilis R. Br.
Amphipogon laguroides R. Br.
Amphipogon turbinatus R. Br.
* *Anthoxanthum odoratum* L.
* *Arrhenantherum bulbosum* (Willd.) C. Presl
* *Arundo donax* L.
* *Avellina michelii* (Savi) Parl.
* *Avena barbata* Link
* *Axonopus compressus* (Swartz.) Beauv.
* *Briza maxima* L.
* *Briza minor* L.
Bromus arenarius Labill.
* *Bromus catharticus* M. Vahl.
* *Bromus hordeaceus* (L.) Pers.
* *Cortaderia selloana* (Schultes et J.H. Schultes) Asch. et Graebn.
* *Cynodon dactylon* (L.) Pers.
* *Cynosurus cristatus* L.
* *Cynosurus echinatus* L.
* *Dactylis glomerata* L.
Danthonia caespitosa Gaud.
Danthonia pilosa R. Br.
Danthonia racemosa R. Br.
Danthonia setacea R. Br. var. *setacea*
Deyeuxia quadriseta Benth.
Dichelachne crinita (L.f.) J.D. Hook.
Dichelachne micrantha (Cav.) Domin
* *Digitaria sanguinalis* (L.) Scop.
Diplopogon setaceus R. Br.
* *Echinochloa crusgalli* (L.) Beauv.
* *Echinopogon ovatus* (G. Forster) P. Beauv.
* *Ehrharta erecta* Lam.
* *Ehrharta calycina* Smith
* *Ehrharta longiflora* Smith
Ehrharta pusilla Nees ex Trin.
* *Ehrharta villosa*
Eragrostis benthamii Mattei
* *Eragrostis curvula* (Schrad.) Nees
Eragrostis elongata (Willd.) Jacq.
* *Festuca arundinacea* Schreber
Festuca littoralis Labill.
Glyceria australis C.E. Hubb.
* *Glyceria maxima* (Hartm.) O.R. Holmberg
* *Hainardia cylindrica* (Willd.) W. Greuter
Hemarthria uncinata R. Br.
* *Holcus lanatus* L.
* *Hordeum glaucum* Steudel
* *Lagurus ovatus* L.
* *Lolium perenne* L.
* *Lolium rigidum* Gaudin
* *Lolium temulentum* L.
* *Melinis minutiflora* P. Beauv.
Microlaena stipoides (Labill.) R. Br.
Neurachne alopecuroidea R. Br.
* *Panicum capillare* L.
* *Panicum maximum* Jacq.
* *Paraphlolis incurva* (L.) C.E. Hubb.
* *Paspalum dilatatum* Poir.
* *Paspalum distichum* L.
* *Paspalum vaginatum* Sw.
Pennisetum clandestinum Hochst. f. ex Chiov.
* *Phleum pratense* L.
* *Phragmites australis* (Cav.) Trin. ex Steud.
Piptatherum miliaceum (L.) Cosson
* *Poa annua* L.
Poa drummondiana Nees
Poa homomalla Nees
Poa poiformis (Labill.) Druce
Poa porphyroclados Nees
* *Poa pratensis* L.
Poa serpentum Nees
* *Polypogon monspeliensis* (L.) Desf.
Polypogon tenellus R. Br.
* *Polypogon viridis* (Gouan) Breistr.
* *Sorghum halepense* (L.) Pers.
Spinifex hirsutus Labill.
* *Sporobolus indicus* (L.) R. Br.
Sporobolus virginicus (L.) Kunth
* *Stenotaphrum secundatum* (Walter) Kunze
Stipa campylachne Nees
Stipa compressa R. Br.
Stipa flavescens Labill.
Stipa hemipogon Benth.

Stipa macalpinei Reader
Stipa mollis R. Br.
Stipa semibarbata R. Br.
Stipa tenuifolia Steud.
Stipa trichophylla Benth.
Tetrarrhena laevis R. Br.
* *Vulpia membranacea* (L.) Dumort.
* *Vulpia myuros* Gmel.

CYPERACEAE
Baumea acuta (Labill.) Palla
Baumea articulata (R. Br.) S.T. Blake
Baumea juncea (R. Br.) Palla
Baumea rubiginosa (Sprengel) Boeckler
Bolboschoenus caldwellii (Cook) Sojak
Carex appressa R. Br.
* *Carex divisa* Hudson
Carex fascicularis Sol. ex Boott
Carex inversa R. Br.
Chorizandra cymbaria R. Br.
Chorizandra enodis Nees
Cyathochaeta avenacea Benth.
Cyathochaeta clandestina (R. Br.) Benth.
* *Cyperus congestus* Vahl
Cyperus laevigatus L.
Cyperus tenellus L.f.
Evandra aristata R. Br.
Evandra pauciflora R. Br.
Fimbristylis velata R. Br.
Gahnia decomposita (R. Br.) Benth.
Gahnia filum
Gahnia deusta (R. Br.) Benth.
Gahnia trifida Labill.
(E) *Gymnoschoenus anceps* (R. Br.) C.B. Clarke
Isolepis cernua (M. Vahl.) Roemer et Schultes
Isolepis congrua Nees
Isolepis cyperoides R. Br.
Isolepis fluitans (L.) R. Br.
Isolepis marginata (Thunb.) A. Dietr.
Isolepis nodosa (Rottb.) R. Br.
Isolepis oldfieldiana (S.T. Blake) K.L. Wilson
* *Isolepis prolifera* (Rottb.) R. Br.
Isolepis stellata (C.B. Clarke) K.L. Wilson
Lepidosperma angustatum R. Br.
Lepidosperma brunonianum Nees
Lepidosperma effusum Benth.
Lepidosperma gladiatum Labill.
Lepidosperma gracile R. Br.
Lepidosperma leptophyllum Benth.
Lepidosperma leptostachyum Benth.
Lepidosperma longitudinale Labill.
(E) *Lepidosperma persecans* S.T. Blake
Lepidosperma squamatum Labill.
Lepidosperma tenue Benth.
Lepidosperma tetraquetrum Nees in Lehm.
Mesomelaena graciliceps (C.B. Clarke) K.L. Wilson
Mesomelaena stygia (R. Br.) Nees
Mesomelaena tetragona (R. Br.) Benth.
Reedia spathacea F. Muell.
Schoenoplectus validus (M. Vahl.) A. Love et D. Love
(?E)*Schoenus acuminatus* R. Br.
Schoenus asperocarpus F. Muell.
Schoenus bifidus (Nees) Boeckler
Schoenus breviculmis Benth.
Schoenus brevisetis (R. Br.) Benth.
Schoenus caespititius W. Fitzg.
(E) *Schoenus cruentus* (Nees) Benth.
Schoenus curvifolius (R. Br.) Benth.
(E) *Schoenus efoliatus* F. Muell.
Schoenus grammatophyllus F. Muell.
Schoenus grandiflorus (Nees) F. Muell.
Schoenus lanatus Labill.
(?E)*Schoenus multiglumis* Benth.
Schoenus nitens (R. Br.) Poiret
Schoenus odontocarpus F. Muell.
Schoenus pleistomoneus F. Muell.
Schoenus rodwayanus W. Fitzg.
Schoenus subbulbosus Benth.
Schoenus sublaxus Kuek.
Schoenus trachycarpus F. Muell.
Tetraria capillaris (F. Muell.) J. Black
Tetraria octandra (Nees.) Kuek.
Tricostularia neesii Lehm. var. *neesii*
Tricostularia neesii var. *elatior* Benth.

ARACEAE
* *Zantedeschia aethiopica* (L.) Sprengel

LEMNACEAE
Lemna disperma Hegelm.

RESTIONACEAE
(E) *Alexgeorgea ganopoda* L. Johnson et Briggs
Anarthria gracilis R. Br.
Anarthria laevis R. Br.
Anarthria prolifera R. Br.
Anarthria scabra R. Br.
Chaetanthus leptocarpoides R. Br.
Ecdeiocolea monostachya F. Muell.
Empodisma gracillimum (F. Muell.) L. Johnson et Cutler
Hypolaena exsulca R. Br.
Hypolaena ramosissima Gilg.
Hypolaena sp.
Lepidobolus sp. aff. *chaetocephalus* (?*preissianus* Nees)
Leptocarpus aristatus R. Br.
Leptocarpus canus Lindley et Nees
Leptocarpus coangustatus Nees
Leptocarpus scariosus R. Br.
Leptocarpus tenax (Labill.) R. Br.
Leptocarpus tenellus (Nees) F. Muell.
Leptocarpus sp. "d"

Leptocarpus sp. "r"
(E) *Leptocarpus* sp. "t"
Lepyrodia drummondiana Steudel
Lepyrodia glauca (Nees) F. Muell.
Lepyrodia heleocharoides Gilg.
Lepyrodia hermaphrodita R. Br.
Lepyrodia muirii F. Muell.
Lepyrodia stricta R. Br.
Loxocarya cinerea R. Br.
Loxocarya fasiculata (R. Br.) Benth.
Loxocarya flexuosa (R. Br.) Benth.
Loxocarya pubescens (R. Br.) Benth.
(E) *Loxocarya* sp. (GK 12794)
Loxocarya aff. *cinerea* "r"
Lyginia barbata R. Br.
Meeboldina denmarkica Suess.
"Pseudoloxocarya grossa" (gen./sp. nov.)
Restio amblycoleus F. Muell.
Restio applanatus Sprengel
Restio "crassus"
Restio gracilior
Restio laxus R. Br.
Restio leptocarpoides Benth.
Restio tremulus R. Br.
Restio ustulatus F. Muell ex Ewart et Sharman

CENTROLEPIDACEAE

Aphelia cyperoides R. Br.
Aphelia drummondii (Hieron.) Benth.
Centrolepis aristata (R. Br.) Roem. et Schult.
(E) *Centrolepis caespitosa* D.A. Cooke
Centrolepis drummondii (Nees) Walp.
Centrolepis fascicularis Labill.
Centrolepis glabra (F. Muell. ex Sond.) Hieron.
Centrolepis humillima F. Muell. ex Benth.
Centrolepis inconspicua W. Fitzg.
Centrolepis mutica (R. Br.) Hieron.
Centrolepis pilosa Hieron.
Centrolepis polygyna (R. Br.) Hieron.
Centrolepis strigosa (R. Br.) Roem. et Schult.

HYDATELLACEAE

Trithuria bibracteata D.A. Cooke
Trithuria submersa J.D. Hook.

XYRIDACEAE

Xyris flexifolia R. Br.
Xyris gracillima F. Muell.
Xyris indivisa Wakef.
Xyris lacera R. Br.
Xyris lanata R. Br.
Xyris laxiflora F. Muell.
(E) *Xyris roycei* Wakef.

PHILYDRACEAE

Philydrella drummondii L.G. Adams
Philydrella pygmaea (R. Br.) Caruel

JUNCACEAE

Juncus amabilis E. Edgar
* *Juncus articulatus* L.
* *Juncus bufonius* L.
Juncus caespiticius E. Meyer
* *Juncus capitatus* Weigel
Juncus gregiflorus L. Johnson
Juncus holoschoenus R. Br.
Juncus kraussii Hochst.
* *Juncus microcephalus* Kunth
* *Juncus oxycarpus* E. Meyer
Juncus pallidus R.Br.
Juncus pauciflorus R. Br.
Juncus planifolius R. Br.
Juncus prismatocarpus R. Br.
Juncus subsecundus Wakef.
Luzula meridionalis Nordensk.

ASPARAGACEAE

* *Myrsiphyllum asparagoides* (L.) Willd.

DASYPOGONACEAE

Acanthocarpus preissii Lehm.
Baxteria australis R. Br.
Calectasia grandiflora Preiss
Chamaexeros serra (Endl.) Benth.
(E) *Chamaexeros* sp. nov.
Dasypogon bromeliifolius R. Br.
Dasypogon hookeri I. Drumm.
Kingia australis R. Br.
Lomandra brittanii T.S. Choo
Lomandra caespitosa (Benth.) Ewart
Lomandra drummondii (F. Muell. ex Benth.) Ewart
Lomandra hastilis (R. Br.) Ewart
Lomandra hermaphrodita (C. Andrews) C. Gardner
Lomandra integra T.D. Macfarlane
Lomandra micrantha (Endl.) Ewart ssp. *micrantha*
Lomandra nigricans T.D. MacFarlane
Lomandra odora (Endl.) Ewart
(E) *Lomandra ordii* F. Muell.
Lomandra pauciflora (R. Br.) Ewart
Lomandra preissii (Endl.) Ewart
Lomandra purpurea (Endl.) Ewart
Lomandra sericea (Endl.) Ewart
Lomandra sonderi (F. Muell.) Ewart
Lomandra suaveolens (Endl.) Ewart

XANTHORRHOEACEAE

Xanthorrhoea gracilis Endl.
Xanthorrhoea preissii Endl.
Xanthorrhoea platyphylla Bedford

PHORMIACEAE

Dianella divaricata R. Br.
Dianella revoluta R. Br. var. *brevicaulis* Ostenf.
Stypandra grandiflora Lindley

ANTHERICACEAE
Agrostocrinum sp. (GJK 1888)
Agrostocrinum scabrum (R. Br.) Baillon
Arthropodium preissii Lehm. ex Endl.
Borya constricta D.M. Churchill
Borya longiscapa D.M. Churchill
Borya nitida Labill.
Borya scirpoidea Lindley
Borya sphaerocephala R. Br.
Caesia micrantha Lindley
Caesia occidentalis R. Br.
Caesia parviflora R. Br.
Chamaescilla corymbosa R. Br. var. *corymbosa*
Chamaescilla corymbosa var. *latifolia* (F. Muell.) R. Henderson
Chamaescilla spiralis (Endl.) F. Muell.
Corynotheca micrantha (Lindl.) Macbride var. *panda* Henderson
Hodgsoniola junciformis (F. Muell.) F. Muell.
Johnsonia acaulis Endl.
Johnsonia lupulina R. Br.
Johnsonia teretifolia Endl.
Laxmannia jamesii G.J. Keighery
Laxmannia minor R. Br.
Laxmannia ramosa Lindley
Laxmannia sessiliflora Decne.
Sowerbaea laxiflora Lindley
Thysanotus arbuscula Baker
Thysanotus arenarius N.H. Brittan
Thysanotus dichotomus (Labill.) R. Br.
(E) *Thysanotus formosus* N.H. Brittan
Thysanotus gracilis R. Br.
(E) *Thysanotus isantherus* R. Br.
Thysanotus manglesianus Kunth
Thysanotus multiflorus R. Br.
Thysanotus patersonii R. Br.
Thysanotus pauciflorus R. Br.
Thysanotus pseudojunceus N.H. Brittan
Thysanotus sparteus R. Br.
Thysanotus tenellus Lindl.
Thysanotus thyrsoideus Baker
Thysanotus triandrus (Labill.) R. Br.
Tricoryne elatior R. Br.
Tricoryne humilis Endl.

ASPHODELACEAE
Bulbine semibarbata (R. Br.) Haw.
* *Trachyandra divaricata* (Jacq.) Kunth.

HYACINTHACEAE
* *Albuca canadensis* (L.) F.M. Leighton

ALLIACEAE
* *Allium ampeloprasum* L.
* *Allium triquetrum* L.
* *Nothoscordum gracile* (Aiton) Stearn

COLCHICACEAE
Burchardia monantha Domin
Burchardia multiflora Lindley
Burchardia umbellata R. Br.
Wurmbea dioica (R. Br.) F. Muell. ssp. *alba* T.D. MacFarlane
Wurmbea monantha (Endl.) T.D. MacFarlane

HAEMODORACEAE
Anigozanthos bicolor Endl. ssp. *decrescens* Hopper
Anigozanthos flavidus Redoute et DC.
Anigozanthos manglesii D. Don ssp. *manglesii*
Anigozanthos preissii Endl.
Anigozanthos viridis Endl. ssp. *viridis*
Conostylis aculeata R. Br. ssp. *aculeata*
Conostylis aculeata R.Br. ssp. *gracilis* Hopper
Conostylis candicans Endl. ssp. *calcicola* Hopper
Conostylis laxiflora Benth.
Conostylis serrulata R. Br.
Conostylis setigera R. Br. ssp. *setigera*
Haemodorum discolor T.D. Macfarlane
Haemodorum laxum R. Br.
Haemodorum paniculatum Lindley
Haemodorum simplex Lindley
Haemodorum sparsiflorum F. Muell.
Haemodorum spicatum R. Br.
Phlebocarya ciliata R. Br.
Tribonanthes australis Endl.
Tribonanthes brachypetala Lindley
Tribonanthes violacea Endl.

AMARYLLIDACEAE
* *Agapanthus praecox* Willd. ssp. *praecox*
* *Amaryllis belladona* L.
* *Narcissus tazetta* L.

HYPOXIDACEAE
Hypoxis glabella R. Br. var. *glabella*
Hypoxis occidentalis Benth. var. *quadriloba* Henderson

IRIDACEAE
* *Babiana stricta* (Ait.) Ker-Gawler
* *Chasmanthe floribunda* (Salisb.) N.E. Br.
* *Crocosmia x crocosmiiflora* (Lemoine) N.E. Br.
* *Ferraria crispa* Burman
* *Freesia leichtlinii* Klatt
* *Gladiolus angustus* L.
* *Gladiolus carneus* Del.
* *Gladiolus undulatus* L.
* *Gynandriris setifolia* (L.f.) R. Foster
* *Homeria flaccida* Sweet
* *Homeria miniata* (Andr.) Sweet
* *Iris germanica* L.
* *Iris unguicularis* Poiret
* *Ixia maculata* L.
* *Ixia paniculata* Del.

Orthrosanthus laxus (Endl.) Benth. var. *laxus*
Orthrosanthus laxus var. *gramineus* (Endl.) Geer.
Orthrosanthus multiflorus Sweet
(E) *Orthrosanthus polystachyus* Benth.
Patersonia babianoides Benth.
Patersonia juncea Lindley
Patersonia limbata Endl.
Patersonia occidentalis R. Br.
Patersonia pygmaea Lindley
Patersonia umbrosa Endl. var. *umbrosa*
Patersonia umbrosa var. *xanthina* (F.Muell.) D.A. Cooke
* *Romulea rosea* (L.) Ecklon
* *Sparaxis bulbifera* (L.) Ker-Gawler
* *Tritonia lineata* (Salisb.) Ker-Gawler
* *Watsonia bulbillifera* J. Mathews
* *Watsonia leipoldtii* L. Bolus
* *Watsonia marginata* (L.f.) Ker Gawler
* *Watsonia versfeldii* J. Mathews et L. Bolus var. *alba* J. Mathews et L. Bolus
* *Watsonia wordsworthiana* J. Mathews et L. Bolus

ORCHIDACEAE
Burnettia forrestii (F. Muell.) Hopper et A.P. Brown
Burnettia nigricans (R. Br.) Hopper et A.P. Brown
(E) *Caladenia abbreviata* Hopper et A.P. Brown
Caladenia applanata Hopper et A.P. Brown ssp. *applanata*
(E) *Caladenia applanata* Hopper et A.P. Brown ssp. *erubescens* Hopper et A.P. Brown
Caladenia arrecta Hopper et A.P. Brown
Caladenia attingens Hopper et A.P. Brown ssp. *attingens*
Caladenia bicalliata R. Rogers
Caladenia brownii Hopper
(E) *Caladenia busselliana* Hopper et A.P. Brown
Caladenia cairnsiana F. Muell.
Caladenia chapmanii Hopper et A.P. Brown
Caladenia citrina Hopper et A.P. Brown
Caladenia corynephora A.S. George
Caladenia ensata Nicholls
(E) *Caladenia evanescens* Hopper et A.P. Brown
Caladenia excelsa Hopper et A.P. Brown
Caladenia ferruginea Nicholls
Caladenia flava R. Br. ssp. *flava*
Caladenia flava R. Br. ssp. *sylvestris* Hopper et A.P. Brown
(E) *Caladenia gardneri* Hopper et A.P. Brown
(E) *Caladenia harringtoniae* Hopper et A.P. Brown
Caladenia heberleana Hopper et A.P. Brown.
Caladenia hirta Lindley ssp. *hirta*
(E) *Caladenia huegelii* N.G. Reichb.
(E) *Caladenia humiliflora* Hopper et A.P. Brown ssp. *meridionalis* Hopper et A.P. Brown
Caladenia infundibularis A.S. George
(E) *Caladenia interjacens* Hopper et A.P. Brown
Caladenia latifolia R. Br.
Caladenia longicauda Lindley ssp. *longicauda*
Caladenia longicauda Lindley ssp. *splendens* Hopper et A.P. Brown
Caladenia longiclavata E. Coleman
Caladenia macrostylis R.D. Fitzg.
(E) *Caladenia magniclavata* Nicholls
Caladenia marginata Lindley
Caladenia nana Endl. ssp. *nana*
Caladenia nana Endl. ssp. *unita* (R.D. Fitzg.) Hopper et A.P. Brown
Caladenia nivalis Hopper et A.P. Brown
Caladenia paludosa Hopper et A.P. Brown
Caladenia pectinata R. Rogers
Caladenia pholcoidea Hopper et A.P. Brown
Caladenia plicata R.D. Fitzg.
Caladenia radiata Nicholls
Caladenia reptans Lindley
Caladenia rhomboidiformis (E. Coleman) M. Clements et Hopper
Caladenia serotina Hopper et A.P. Brown
Caladenia viridescens Hopper et A.P. Brown
(E) *Caladenia winfieldii* Hopper et A.P. Brown
Calochilus robertsonii Benth.
Corybas despectans D.L. Jones et R.C. Nash
Corybas recurvus D. Jones
Corybas abditus D. Jones
Cryptostylis ovata R. Br.
Cyanicula deformis (R. Br.) Hopper et A.P. Brown
Cyanicula gemmata (Lindley) Hopper et A.P. Brown
Cyanicula gertrudeae (Ostenf.) Hopper et A.P. Brown
Cyanicula sericea (Lindley) Hopper et A.P. Brown
Cyrtostylis huegelii Endl.
Cyrtostylis robusta D. Jones et M. Clements
Cyrtostylis tenuissima (Nicholls et Goadby) D. Jones et M. Clements
Diuris aff. *amplissima* D. Jones
Diuris carinata Lindley
Diuris drummondii Lindley
Diuris emarginata R. Br.
Diuris filifolia Lindley
Diuris heberlei D. Jones
Diuris laevis R.D. Fitzg.
Diuris laxiflora Lindley
Diuris longifolia R. Br.
Diuris pauciflora R. Br.
Diuris setacea R. Br.
Drakaea glyptodon R.D. Fitzg.
Drakaea livida J. Drummond
Drakaea micrantha Hopper et A.P. Brown
Drakaea thynniphila A.S. George
Elythranthera brunonis (Endl.) A.S. George
Elythranthera emarginata (Lindley) A.S. George
Epiblema grandiflorum R. Br.
Eriochilus dilatatus Lindley ssp. *dilatatus*
Eriochilus dilatatus Lindley ssp. *multiflorus* (Lindley) Hopper et A.P. Brown

Eriochilus dilatatus Lindley ssp. *magnus* Hopper et A.P. Brown
Eriochilus helonomos Hopper et A.P. Brown
(E) *Eriochilus pulchellus* Hopper et A.P. Brown
Eriochilus scaber Lindley ssp. *scaber*
(E) *Eriochilus scaber* Lindley ssp. *orbifolia* Hopper et A.P. Brown
(E) *Eriochilus valens* Hopper et A.P. Brown
Eriochilus tenuis Lindley
Gastrodia lacista D. Jones
Leporella fimbriata (Lindley) A.S. George
Leptoceras menziesii (R. Br.) Lindley
Lyperanthus serratus Lindley
Microtis alba R. Br.
Microtis atrata Lindley
Microtis brownii H.G. Reichb.
(E) *Microtis familiaris* R. Bates
(E) *Microtis globula* R. Bates
Microtis media R. Br. ssp. *media*
Microtis media ssp. *densiflora* (Benth.) R. Bates
Microtis media ssp. *quadrata* R. Bates
Microtis orbicularis R. Rogers
(E) *Microtis pulchella* R. Br.
Microtis aff. *unifolia* (G. Forster) H.G. Reichb.
* *Monadenia bracteata* (Sw.) Dur. et Schinz.
Paracaleana linearifolia Hopper et A.P. Brown
Paracaleana nigrita (Lindley) Blaxell
Praecoxanthus aphyllus (Benth.) Hopper et A.P. Brown
Prasophyllum brownii H.G. Reichb.
Prasophyllum calcicola R. Bates
Prasophyllum cyphochilum Benth.
Prasophyllum drummondii H.G. Reichb.
Prasophyllum elatum R. Br.
Prasophyllum fimbria H.G. Reichb.
Prasophyllum gibbosum R. Br.
(E) *Prasophyllum* aff. *gibbosum* R. Br.
Prasophyllum giganteum Lindley
Prasophyllum hians H.G. Reichb.
Prasophyllum macrostachyum R. Br.
Prasophyllum odoratum R. Rogers
Prasophyllum parvifolium Lindley
Prasophyllum aff. *parvifolium* Lindley
Prasophyllum plumaeforme R.D. Fitzg.
Prasophyllum regium R. Rogers
Prasophyllum triangulare R.D. Fitzg.
Pterostylis aspera D. Jones et M. Clements
Pterostylis barbata Lindley
Pterostylis aff. *dilatata* A.S. George
Pterostylis aff. *nana* R. Br.
Pterostylis aff. *plumosa* Cady
Pterostylis pyramidalis Lindley
Pterostylis recurva Benth.
Pterostylis rogersii E. Coleman
(?E)*Pterostylis turfosa* Lindley
Pterostylis vittata Lindley
Pterostylis aff. *vittata* Lindley
Rostranthus forrestii (F. Muell.) Hopper et A.P. Brown
Rostranthus nigricans (R. Br.) Hopper et A.P. Brown
Thelymitra antennifera (Lindley) J.D. Hook.
Thelymitra benthamiana H.G. Reichb.
Thelymitra canaliculata R. Br.
Thelymitra cornicina H.G. Reichb.
Thelymitra crinita Lindley
Thelymitra cucullata Rupp.
Thelymitra flexuosa Endl.
Thelymitra fuscolutea R. Br.
Thelymitra aff. *holmesii* Nicholls
(E) *Thelymitra jacksonii* Hopper et A.P. Brown
Thelymitra aff. *longifolia* J. Forst. et G. Forst.
Thelymitra mucida Fitzg.
Thelymitra aff. *nuda* R. Br.
Thelymitra pauciflora R. Br.
Thelymitra spiralis (Lindley) F. Muell.
Thelymitra tigrina R. Br.
Thelymitra variegata (Lindley) F. Muell.
Thelymitra villosa Lindley

DICOTYLEDONS

CASUARINACEAE

Allocasuarina decussata (Benth.) L. Johnson
Allocasuarina fraseriana (Miq.) L. Johnson
Allocasuarina huegeliana (Miq.) L. Johnson
Allocasuarina humilis (Otto et Dietr.) L. Johnson
Allocasuarina lehmanniana (Miq.) L. Johnson
Allocasuarina trichodon (Miq.) L. Johnson
Allocasuarina thuyoides (Miq.) L. Johnson

URTICACEAE

Parietaria debilis G. Forst.
* *Soleirolia soleirolii* (Req.) Dandy
* *Urtica urens* L.

PROTEACEAE

Adenanthos apiculatus R. Br.
Adenanthos barbigerus Lindley
Adenanthos cuneatus Labill.
Adenanthos cunninghamii Meissner
(E) *Adenanthos detmoldii* F. Muell.
Adenanthos meisneri Lehm.
Adenanthos obovata Labill.
Adenanthos sericeus Labill. ssp. *sericeus*
Banksia attenuata R. Br.
Banksia gardneri A.S. George
Banksia goodii R. Br.
Banksia grandis R. Br.
Banksia ilicifolia R. Br.
Banksia littoralis R. Br.
Banksia meisneri Lehm. var. *ascendens* A.S. George
(E) *Banksia occidentalis* R. Br. ssp. *formosa* Hopper

Banksia occidentalis R. Br. ssp. *occidentalis*
Banksia praemorsa Andrews
Banksia quercifolia R. Br.
Banksia seminuda (A.S. George) B. Rye ssp. *seminuda*
(E) Banksia seminuda ssp. remanens Hopper
Banksia verticillata R. Br.
Conospermum acerosum Lindley
Conospermum caeruleum R. Br.
Conospermum capitatum R. Br.
Conospermum debile
Conospermum flexuosum R. Br.
Conospermum floribundum Benth.
Conospermum petiolare R. Br.
Conospermum teretifolium R. Br.
Conospermum triplinervium R. Br.
(E) *Conospermum* sp.
Dryandra armata R. Br.
Dryandra baxteri R. Br.
Dryandra bipinnatifida R. Br.
Dryandra cuneata R. Br.
Dryandra formosa R. Br.
Dryandra mucronulata R. Br.
Dryandra nivea (Labill.) R. Br.
Dryandra serra R. Br.
Dryandra sessilis (Knight) Domin
Dryandra subpinnatifida C. Gardner
(E) *Grevillea brachystylis* Meissner var. *australis* Keighery
Grevillea brownii Meissner
Grevillea cirsiifolia Meissner
Grevillea diversifolia Meissner ssp. *subterisericata* MacGillivray
Grevillea drummondii Meissner ssp. *centristigma* MacGillivray
Grevillea fasiculata R. Br.
Grevillea fuscolutea Keighery
Grevillea manglesioides Meissner ssp. *manglesioides*
Grevillea manglesioides ssp. *papillosa* MacGillivray
Grevillea occidentalis R. Br.
Grevillea pilulifera (Lindley) Druce
Grevillea pulchella (R. Br.) Meissner
Grevillea quercifolia R. Br.
Grevillea pulchella (R. Br.) Meissner
Grevillea ripicola A.S. George
Grevillea trifida (R. Br.) Meissner
Franklandia fucifolia R. Br.
Hakea amplexicaulis R. Br.
Hakea ceratophylla (Sm.) R. Br.
Hakea cyclocarpa Lindley
Hakea elliptica (Smith) R. Br.
Hakea falcata R. Br.
Hakea florida R. Br.
Hakea lasiantha R. Br.
Hakea lasianthoides B.L. Rye
Hakea linearis R. Br.
Hakea lissocarpha R. Br.
Hakea oleifolia (Smith) R. Br.
Hakea prostrata R. Br.
Hakea ruscifolia Labill.
Hakea suaveolens R. Br.
Hakea sulcata R. Br.
Hakea trifurcata (Sm.) R. Br.
Hakea undulata R. Br.
Hakea varia R. Br.
Isopogon attenuatus R. Br. var. *attenuatus*
Isopogon axillaris R. Br.
Isopogon buxifolius R. Br.
Isopogon formosus R. Br.
Isopogon uncinatus R. Br.
Isopogon sphaerocephalus Lindley
Isopogon teretifolius R. Br.
Lambertia inermis R. Br.
Lambertia orbifolia C. Gardner
Lambertia propinqua R. Br.
Lambertia uniflora R. Br.
Persoonia graminea R. Br.
Persoonia elliptica R. Br.
Persoonia longifolia R. Br.
Persoonia microcarpa R. Br.
Persoonia saccata R. Br.
Persoonia teretifolia
Petrophile acicularis R. Br.
Petrophile divaricata R. Br.
Petrophile diversifolia R. Br.
Petrophile linearis R. Br.
Petrophile longifolia R. Br.
Petrophile media
Petrophile rigida R. Br.
Petrophile serruriae R. Br.
Petrophile squamata R. Br. ssp. "A" (short-leaved form)
Petrophile squamata R. Br. ssp. "B" (fine-leaved form)
Stirlingia latifolia (R. Br.) Steudel
Stirlingia simplex Lindley
Stirlingia tenuifolia (R. Br.) Steudel
Strangea stenocarpoides (F. Muell. ex Benth.) C. Gardner
Synaphea favosa R. Br.
Synaphea gracillima Lindley
Synaphea petiolaris R. Br.
Synaphea polymorpha R. Br.
Synaphea preissii Meissner
Synaphea reticulata (Smith) C. Gardner
Xylomelum occidentale R. Br.

SANTALACEAE
(E) *Choretrum lateriflorum* R. Br.
Exocarpos odoratus (Miq.) A.DC.
Exocarpos sparteus R. Br.
Leptomeria cunninghamii Miq.
Leptomeria lehmaniannii Miq.

Leptomeria ericoides Miq.
Leptomeria pauciflora R. Br.
Leptomeria scrobiculata R. Br.
Leptomeria squarrulosa R. Br.
Leptomeria spinosa (Miq.) A.DC.

OLACACEAE
Olax phyllanthi (Labill.) R. Br.
Olax benthamiana Miq.

LORANTHACEAE
Amyema miquelii (Lehm. ex Miq.) Tiegh.
Nuytsia floribunda (labill.) R. Br.

POLYGONACEAE
* *Emex australis* Steinh.
* *Fagopyrum esculentum* Moench.
Muehlenbeckia appressa (Labill.) Meissner
* *Polygonum attenuatum* R. Br.
Polygonum hydropiper L.
Polygonum prostratum R. Br.
Polygonum salicifolium Brouss. ex Willd.
* *Rumex acetosella* L.
Rumex brownii Campdera
* *Rumex conglomeratus* Murr.
* *Rumex crispus* L.
* *Rumex frutescens* Thouars
* *Rumex pulcher* L. ssp. *pulcher*

CHENOPODIACEAE
Atriplex bunburyana F. Muell.
Atriplex cinerea Poiret
* *Atriplex hortensis* L.
Atriplex hypoleuca Nees
Atriplex isatidea Moq.
Atriplex paludosa R. Br. ssp. *baudinii* (Moq.) Aellen
* *Atriplex prostrata* Boucher ex DC
* *Chenopodium album* L.
* *Chenopodium ambrosioides* L.
* *Chenopodium glaucum* L.
* *Chenopodium murale* L.
Chenopodium pumilio R. Br.
Halosarcia halocnemoides (Nees) P.G. Wilson ssp. *halocnemoides*
Halosarcia indica (Willd.) P.G. Wilson ssp. *bidens* (Nees) P.G. Wilson
Halosarcia pergranulata (J.M. Black) P.G. Wilson ssp. *pergranulata*
Halosarcia pterygosperma (J.M. Black) P.G. Wilson ssp. *pterygosperma*
Halosarcia syncarpa P.G. Wilson
Maireana brevifolia (R. Br.) P.G. Wilson
Maireana oppositifolia (F. Muell.) P.G. Wilson
Rhagodia baccata (Labill.) Moq. ssp. *baccata*
Rhagodia bacatta (Labill.) Moq. ssp. *dioica*
Rhagodia crassifolia R. Br.
Salsola kali L. ssp. kali
Sarcocornia blackiana (Ulbr.) A.J. Scott
Sarcocornia quinqueflora (Bunge ex Ung-Sternb.) A.J. Scott ssp. *quinqueflora*
Sarcocornia blackiana (Ulbr.) A.J. Scott
Suaeda australis (R. Br.) Moq.
Threlkeldia diffusa R. Br.

AMARANTHACEAE
Alternanthera nodiflora R. Br.
* *Amaranthus albus* L.
Hemichroa diandra R. Br.
Ptilotus declinatus Nees
Ptilotus sericostachyus (Nees) F. Muell.
Ptilotus stirlingii (Lindley) F. Muell. var. *stirlingii*
Ptilotus stirlingii var. *laxus* (Benth.) Benl.

GYROSTEMONACEAE
Gyrostemon sheathii W.V. Fitzg.
Gyrostemon thesioides (J.D. Hook.) A.S. George

PHYTOLACCACEAE
* *Phytolacca octandra* L.

AIZOACEAE
* *Carpobrotus aequilaterus* (Haw.) N.E. Br.
* *Carpobrotus edulis* (L.) L. Bolus
Carpobrotus virescens (Haw.) Schwantes
* *Drosanthemum candens* (Haw.) Schwantes
* *Lampranthus glaucus* (L.) N.E. Br.
* *Mesembryanthemum crystallinum* L.
* *Tetragonia decumbens* Thunb.
Tetragonia implexicoma (Miq.) J.D. Hook.
Tetragonia tetragonioides (Pallas) Kuntze

PORTULACACEAE
Calandrinia brevipedata F. Muell.
Calandrinia calyptrata Hook.
Calandrinia corrigioloides F. Muell. ex Benth.
Calandrinia granulifera Benth.
Calandrinia liniflora Fenzl

CARYOPHYLLACEAE
* *Cerastium glomeratum* Thuill.
* *Cerastium semidecandrum* L.
* *Corrigiola littoralis* L.
* *Lychnis coronaria* L.
* *Moenchia erecta* (L.) P. Gaertner
* *Petrohagia velutina* (Guss.) P. Ball et Heyw.
* *Polycarpon tetraphyllum* (L.) L.
* *Sagina maritima* Don
* *Sagina procumbens* L.
* *Silene gallica* L. var. *gallica*
* *Silene gallica* L. var. *quinquevulnera* (L.) Mert. et Koch
* *Silene nocturna* L.
* *Spergula diandra* Heldr. et Sart
* *Stellaria media* (L.) Vill.
* *Vaccaria pyramidata* L.

RANUNCULACEAE

Clematis microphylla DC.
Clematis pubescens Huegel ex Endl.
Ranunculus colonorum Endl.
* *Ranunculus muricatus* L.
Ranunculus rivularis Banks et Sol. ex DC.

LAURACEAE

Cassytha glabella R. Br. forma *casuarinae* (Nees) J.Z. Weber
Cassytha flava Nees
Cassytha melantha R. Br.
Cassytha micrantha Meisn.
Cassytha pomiformis Nees
Cassytha racemosa Nees forma *racemosa*
Cassytha racemosa forma *pilosa* (Benth.) J.Z. Weber

FUMARIACEAE

* *Fumaria capreolata* L.
* *Fumaria muralis* Sond. ex Koch
* *Fumaria officinalis* L.

BRASSICACEAE

* *Brassica rapa* L. ssp. *sylvestris* (L.) Janden
* *Brassica tournefortii* Gouan
* *Cakile maritima* Scop.
* *Capsella bursa-pastoris* (L.) Medikus
* *Cardamine hirsuta* L.
Cardamine paucijuga Turcz.
* *Coronopus didymus* (L.) Smith
* *Diplotaxis muralis* (L.) DC.
* *Heliophila pusilla* L.f.
* *Hirschfieldia incana* (L.) Lag-Foss.
* *Hymenolobus procumbens* (L.) Nutt.
* *Lepidium africanum* (Burm.) DC.
Lepidium foliosum Desv.
Lepidium linifolium (Desv.) Steudel
Lepidium pseudohyssopifolium Hewson
Lepidium rotundum (Desv.) DC.
* *Nasturtium officinale* R. Br. in Ait.
* *Rapistrum rugosum* (L.) All
* *Raphanus raphanistrum* L.
* *Rorippa dictyosperma* (F. Muell.) L. Johnson
* *Sinapis arvensis* L.
* *Sisymbrium irio* L.
* *Sisymbrium orientale* L.
(E?)*Stenopetalum robustum* Endl.

RESEDACEAE

* *Reseda luteola* L.

DROSERACEAE

Drosera bulbosa Hook.
Drosera dichrosepala Turcz.
Drosera erythrorhiza Lindley ssp. *erythrorhiza*
Drosera erythrorhiza Lindley ssp. *squarrosa*
Drosera gigantea Lindley
Drosera glanduligera Lehm.
(E) *Drosera hamiltonii* C.R.P. Andrews
Drosera huegelii Endl.
Drosera macrantha Endl. ssp. *macrantha*
Drosera menziesii R. Br. ssp. *menziesii*
Drosera microphylla Endl.
Drosera modesta Diels
(E) *Drosera myriantha* Planchon
Drosera neesii Lehm. ssp. *neesii*
Drosera occidentalis Morrison ssp. *australis*
Drosera omissa Diels
Drosera pallida Lindley
Drosera platypoda Turcz.
Drosera platystigma Lehm.
Drosera pulchella Lehm.
Drosera pygmaea DC.
Drosera ramellosa Lehm.
Drosera stolonifera Endl. ssp. *stolonifera*
Drosera subhirtella Planchon

CRASSULACEAE

* *Aeonium castello-pavoniae* Bolle
* *Cotyledon orbiculare* L.
Crassula colorata (Nees) Ostenf. var. *colorata*
Crassula colorata var. *acuminata* (Reader) Toelken
Crassula decumbens Thunb. var. *decumbens*
* *Crassula glomerata* P. Bergius
* *Crassula natans* Thunb. var. *minus* (Eckl. et Zeyh.) Rowley
Crassula sieberiana (J.A. et J.H. Schultes) Druce ssp. *tetramera* Tolken
* *Crassula thunbergiana* J.A. Schultes ssp. *thunbergiana*
* *Crassula tetragona* L. ssp. *robusta* (Toelken) Toelken

CEPHALOTACEAE

Cephalotus follicularis Labill.

SAXIFRAGACEAE

Eremosyne pectinata Endl.

PITTOSPORACEAE

Billardiera candida (Huegel ex Endl.) E.M. Bennett
Billardiera coerulea-punctata (Klotzsch) E.M. Bennett
Billardiera drummondiana (Putterl.) E.M. Bennett var. *drummondiana*
Billardiera erubescens (Putterl.) E.M. Bennett
Billardiera floribunda (Putterl.) F. Muell.
Billardiera laxiflora (Benth.) E.M. Bennett
Billardiera parviflora DC. var. *parviflora*
Billardiera sericea (Turcz.) E.M. Bennett
Billardiera variifolia DC.
Cheiranthera preissiana Putterl. var. *planifolia* E.M. Bennett
(E) *Sollya drummondii* Morren
Sollya heterophylla Lindley

ROSACEAE

* *Acaena echinata* Nees var. *retrorsumpilosa* (Bitter) Orch.
* *Acaena novae-zelandiae* Kirk
* *Cotoneaster glaucophyllus* L.
* *Rosa chinensis* Jacq. x R. multiflora Thunb. ex Murray
* *Rosa rubiginosa* L.
* *Rubus discolor* Weihe et Nees
* *Rubus selmeri* Lindeb. ex F. Aresch.
* *Rubus ulmifolius* Schott
* *Sanguisorba minor* Scop.

MIMOSACEAE

Acacia acuminata Benth.
Acacia acutifolia Maiden et Blakely
Acacia alata R. Br.
Acacia assimilis S. Moore
Acacia ataxiphylla Benth.
Acacia baxteri Benth.
Acacia biflora R. Br.
Acacia browniana Wendl. var. *browniana*
Acacia browniana Wendl. var. *endlicheri*
Acacia browniana Wendl. var. *obscura* (A.DC.) Maslin
Acacia cochlearis (Labill.) H. Wendl.
Acacia costata Benth.
Acacia crassiuscula Wendl.
Acacia crispula Benth.
Acacia cyclops Cunn. ex Don
* *Acacia decurrens* (Wendl.) Willd.
Acacia divergens Benth.
* *Acacia dealbata* Link
Acacia drummondii Lindley ssp. *elegans* B.R. Maslin
Acacia extensa Lindley
Acacia fragilis Maiden et Blakely
Acacia gilbertii Meissner
Acacia hastulata Smith in Rees
Acacia huegelii Benth.
Acacia incurva Benth.
Acacia insolita E. Pritzel
Acacia laricina Meissner
Acacia latipes Benth.
Acacia leioderma Maslin
Acacia lateriticola Maslin
Acacia littorea Maslin
Acacia luteola Maslin
Acacia microbotrya Benth.
Acacia mooreana W.V. Fitzg.
Acacia multispicata Benth.
* *Acacia melanoxylon* R. Br.
Acacia myrtifolia (Smith) Willd.
Acacia nervosa DC.
Acacia obovata Benth.
Acacia paradoxa DC.
(E) *Acacia pentadenia* Lindley
(E) *Acacia* aff. *pentadenis* (Annels 3700)
Acacia preissiana (Meissner) Maslin
Acacia prismifolia E. Pritzel
Acacia pulchella R. Br. var. *pulchella*
Acacia pulchella R. Br. var. *glaberrima* Meissner
Acacia pulchella R. Br. var. *goadbyi* (Domin.) Maslin
* *Acacia pycnantha* Benth.
Acacia pycnocephala Maslin
Acacia robiniae Maslin
Acacia rostellifera Benth.
Acacia saligna (Labill.) H.L. Wendl.
(E) *Acacia scalpelliformis* Meissner
Acacia semitrullata Maslin
Acacia subcaerulea Lindley
(E) *Acacia subracemosa* Maslin
Acacia sulcata R. Br.
(E) *Acacia tayloriana* F. Muell.
Acacia tetragonocarpa Meissner
Acacia triptycha F. Muell. ex Benth.
Acacia uliginosa Maslin
Acacia urophylla Benth. ex Lindley
Acacia varia Maslin var. *varia*
Acacia willdenowiana H.L. Wendl.
Paraserianthes lophantha (Willd.) I. Nielsen

CAESALPINIACEAE

Labichea punctata Benth. in Lindley

PAPILIONACEAE

Aotus carinata Meissner
Aotus sp. aff. *diffusa*
Aotus genistoides Turcz.
Aotus gracillima Meissner
Aotus intermedia Meissner
(E) *Aotus passerinoides* Meissner
Aotus procumbens Meissner
Aotus villosa
Bossiaea aquifolium Benth.
Bossiaea dentata (R. Br.) Benth.
(E) *Bossiaea disticha* Lindley
Bossiaea eriocarpa Benth.
Bossiaea laidlawiana Tovey et P. Morris
Bossiaea linophylla R. Br.
Bossiaea ornata (Lindley) Benth.
Bossiaea rufa R. Br.
Bossiaea webbii F. Muell.
Brachysema praemorsum Meissner
Brachysema sericeum (Sm.) Domin.
Burtonia conferta DC.
Burtonia scabra R. Br.
Burtonia villosa Meissner
Chorizema aciculare (DC.) C. Gardner
Chorizema cordatum Lindley
Chorizema ilicifolium Labill.
Chorizema diversifolium DC.
Chorizema glycinifolium (Sm.) Druce
Chorizema reticulatum Meissner

Chorizema rhomboideum R. Br.
(E) *Chorizema* sp aff. *varium* (Annels 2189)
* *Cytisus prolifer* L.f.
Daviesia alternifolia Endl.
Daviesia angulata Benth.
Daviesia benthamii Meissner
Daviesia brachyphylla M.D. Crisp
Daviesia colletoides
Daviesia cordata Smith
Daviesia decurrens Meissner
Daviesia divaricata
Daviesia flexuosa Benth.
Daviesia gracilis M.D. Crisp
Daviesia horrida Preiss ex Meissner
Daviesia incrassata Smith
Daviesia inflata M.D. Crisp
Daviesia longifolia Benth.
Daviesia oppositifolia Endl.
Daviesia polyphylla Benth.
Daviesia preissii Meissner
Dillwynia cinerascens R. Br. ex Sims
Dillwynia uncinata (Turcz.) J. Black
* *Dipogon lignosus* (L.) Verdc.
Euchilopsis linearis (Benth.) F. Muell.
Eutaxia densifolia Turcz.
Eutaxia epacridoides Meissner
Eutaxia obovata (Labill.) C. Gardner
Eutaxia parvifolia Benth.
Eutaxia virgata Benth.
Gastrolobium bilobum R. Br.
Gastrolobium brownii Meissner
Gastrolobium callistachys Meissner
Gastrolobium forrestii Ewart
Gastrolobium villosum Benth.
* *Genista canariensis* L.
* *Genista linifolia* L.
Gompholobium amplexicaule Meissner
Gompholobium aristatum Benth.
Gompholobium burtonioides Meissner
Gompholobium capitatum A. Cunn.
Gompholobium knightianum Lindley
Gompholobium marginatum R. Br.
Gompholobium ovatum Meissner
Gompholobium polymorphum R. Br.
Gompholobium tomentosum Labill.
Gompholobium venustum R. Br.
Goodia lotifolia Salisb.
Hardenbergia comptoniana (Andrews) Benth.
Hovea chorizemifolia (Sweet) DC.
Hovea stricta
Hovea trisperma Benth.
Hovea elliptica (Sm.) DC.
Isotropis cuneifolia (Sm.) Domin.
Jacksonia alata Benth.
Jacksonia aphylla (Turcz.) Druce
Jacksonia furcellata (Bonpl.) DC.
Jacksonia horrida DC.
Jacksonia mollissima W. Fitzg.
Jacksonia spinosa (Labill.) R. Br.
Jacksonia sternbergiana Huegel
(E) *Jansonia formosa* Kipp. ex Lindley
Kennedia carinata (Benth.) Domin.
Kennedia coccinea Vent.
(E) *Kennedia glabrata* (Benth.) Lindley
(E) *Kennedia macrophylla* (Meissner) Benth.
Kennedia microphylla Meissner
Kennedia prostrata R. Br.
Kennedia stirlingii Lindley
Latrobea brunonis (Benth.) Meissner
Latrobea diosmifolia Benth.
Latrobea genistoides (Meissner) Benth.
Latrobea hirtella (Turcz.) Benth.
Latrobea tenella (Meissner) Benth. var. *tenella*
* *Lathyrus tingitanus* L.
* *Lathyrus sylvestris* L.
* *Lotus angustissimus* L.
* *Lotus suaveolens* Pers.
* *Lotus uliginosus* Schkuhr
* *Lupinus luteus* L.
* *Medicago arabica* (L.) Hudson
* *Medicago lupulina* L.
* *Medicago polymorpha* L. var. *brevispina* (Benth.) Heyn
* *Meliotus indica* (L.) All.
Mirbelia dilatata R. Br.
Mirbelia ovata Meissner
Mirbelia spinosa Benth.
Nemcia hookeri (Meissner) M.D. Crisp
Nemcia spathulata (Benth.) M.D. Crisp
* *Ornithopus compressus* L.
* *Ornithopus pinnatus* (Miller) Druce
Oxylobium carinatum (Meissner) Benth.
Oxylobium coriaceum (Sm.) C. Gardner
Oxylobium drummondii Meissner
Oxylobium lanceolatum (Vent.) Druce
Oxylobium linearifolium (G. Don.) Domin.
Oxylobium spathulatum (Meissner) Benth.
Phyllota barbata Benth.
* *Psoralea pinnata* L.
Pultenaea adunca Turcz.
Pultenaea aspalathoides Meisn.
Pultenaea barbata C. Andrews
Pultenaea ?calycina
Pultenaea drummondii Meissner
Pultenaea ericifolia Benth.
Pultenaea ochreata Meissner
Pultenaea pinifolia Meissner
Pultenaea reticulata (Sm.) Benth.
Pultenaea skinneri F. Muell.
Pultenaea strobilifera Meissner
Pultenaea verruculosa Turcz.
Pultenaea vestita R. Br.
Sphaerolobium alatum Benth.
Sphaerolobium fornicatum Benth.

Sphaerolobium grandiflorum (R. Br.) Benth.
Sphaerolobium linophyllum (Huegel) Benth.
Sphaerolobium macranthum Meissner
Sphaerolobium medium R. Br.
Sphaerolobium nudiflorum (Meissner) Benth.
Sphaerolobium racemulosum Benth.
Sphaerolobium scabriusculum Meissner
Sphaerolobium vimineum Sm.
Templetonia retusa (Vent.) R. Br.
* *Trifolium arvense* L.
* *Trifolium campestre* Schreber
* *Trifolium cernuum* Brot.
* *Trifolium dubium* Sibth.
* *Trifolium glomeratum* L.
* *Trifolium hirtum* All.
* *Trifolium ligusticum* Balbis ex Lois.
* *Trifolium repens* L.
* *Trifolium striatum* L.
* *Trifolium tomentosum* L.
* *Ulex europaeus* L.
* *Vicia hirsuta* (L.) Gray
* *Vicia sativa* L. ssp. *sativa*
* *Vicia sativa* ssp. *nigra* (L.) Ehrh.
Viminaria juncea (Schrader et Wendl.) Hoffsgg.

GERANIACEAE

* *Erodium botrys* (Cav.) Bertol
* *Erodium cicutarium* (L.) L'. Her.
Erodium cygnorum Nees in Lehm. ssp. *cygnorum*
* *Eranium dissectum* L.
Geranium drummondii Carolin
Geranium retrorsum L.Her. ex DC.
Geranium solanderi Carolin
Pelargonium alchemilloides (L.) L. Her. ssp. *alchemilloides*
Pelargonium australe Willd.
* *Pelargonium capitatum* (L.) L.Her. ex Ait.
Pelargonium drummondii Turcz.
Pelargonium littorale Huegel

OXALIDACEAE

* *Oxalis corniculata* L.
Oxalis perennans Haw.
* *Oxalis flava* L.
* *Oxalis incarnata* L.
* *Oxalis pes-caprae* L.
* *Oxalis polyphylla* Jacq.
* *Oxalis purpurea* L.

LINACEAE

Linum marginale Cunn. ex Planchon
* *Linum trigynum* L.

ZYGOPHYLLACEAE

Nitraria billardieri DC.

RUTACEAE

Asterolasia pallida Benth.
Asterolasia squamuligera Hook.
Boronia alata Sm.
Boronia albiflora R. Br. ex Benth.
Boronia crenulata Sm. var. *crenulata*
Boronia crenulata var. *pubescens* Benth.
Boronia crassipes Bartling
Boronia denticulata Smith
Boronia dichotoma Lindley
Boronia fastigata Bartling
Boronia gracilipes F. Muell.
Boronia heterophylla F. Muell.
Boronia juncea Bartling
Boronia megastigma Nees ex Bartling
Boronia molloyae J. Drumm.
Boronia pulchella Turcz.
Boronia spathulata Lindley
Boronia stricta Bartling
Boronia subsessilis Benth.
Boronia virgata P.G. Wilson
Boronia sp. aff. *juncea*
Boronia sp. aff. *spathulata*
Chorilaena quercifolia Endl.
Crowea angustifolia Smith var. *angustifolia*
Crowea angustifolia var. *dentata* (Benth.) P.G. Wilson
Diplolaena dampieri Desf.
Diplolaena drummondii (Benth.) Ostenf.
Diplolaena microcephala Bartling var. *microcephala*
Eriostemon nodiflorus Lindley var. *nodiflorus*
Eriostemon spicatus A. Rich.
Phebalium anceps DC.
Phebalium rude Bartl. ssp. *rude*

TREMANDRACEAE

Platytheca galioides Steetz
Tetratheca affinis Endl.
(E) *Tetratheca elliptica* J. Thompson
(E) *Tetratheca filiformis* Benth.
Tetratheca hirsuta Lindley
Tetratheca hispidissima Steetz
Tetratheca setigera Endl.
Tremandra diffusa R. Br.
Tremandra stelligera R. Br.

POLYGALACEAE

Comesperma calymega Labill.
Comesperma ciliatum Steetz.
Comesperma confertum Labill.
Comesperma flavum DC.
Comesperma nudiusculum DC.
Comesperma virgatum Labill.
Comesperma volubile Labill.
* *Polygala myrtifolia* L.
* *Polygala virgata* Thunb.

EUPHORBIACEAE

Adriana quadripartita (Labill.) Gaudich
Amperea ericoides Adr. Juss.

Amperea micrantha Benth.
(E) *Ampera protensa* Nees
(E) *Amperea volubilis* F. Muell. ex Benth.
Ampera sp. (CJR 227)
Beyeria viscosa (Labill.) Miq.
* *Euphorbia peplus* L.
* *Euphorbia paralias* L.
* *Euphorbia helioscopia* L.
* *Mercurialis annua* L.
Monotaxis grandiflora Endl.
Monotaxis occidentalis Endl.
Phyllanthus calycinus Labill.
Phyllanthus sp. (?*scaber* Klotzsch)
Poranthera huegelii Klotzch
Poranthera microphylla Brongn.
Ricinocarpus glaucus Endl.
* *Riccinus communis* L.

CALLITRICHACEAE

* *Callitriche stagnalis* Scop.

STACKHOUSIACEAE

Stackhousia pubescens Labill.
Tripterococcus brunonis Endl.
(E) *Tripterococcus* sp. nov. (CJR 414)

SAPINDACEAE

Dodonaea aptera Miq.
Dodonaea ceratocarpa Endl.
Dodonaea trifida F. Muell.
Dodonaea viscosa Jacq. ssp. *spatulata* (Smith) J.G. West

RHAMNACEAE

Cryptandra arbutiflora Fenzl
Cryptandra pungens Steudel
Cryptandra tubulosa Fenzl
Pomaderris myrtilloides Fenzl
Spyridium globulosum (Labill.) Benth.
(E) *Spyridium spadiceum* (Fenzl) Benth.
Trymalium floribundum Steud.
(E) *Trymalium* aff. *floribundum* (R.D. Royce 4286)
Trymalium ledifolium Fenzl var. *ledifolium*

MALVACEAE

* *Lavatera arborea* L.
Lavatera plebeia Sims var. *plebeia*
Lavatera plebeia var. *tomentosa* Hook. f.
* *Malva parviflora* L.
* *Modiola caroliniana* (L.) G. Don.
Sida hookeriana Miq.

STERCULIACEAE

Guichenotia ledifolia Gay
Lasiopetalum cordifolium Endl.
(E) *Lasiopetalum floribundum* Benth. ssp. nov.
Rulingia corylifolia R.A. Graham
Rulingia cygnorum (Steud.) C. Gardner var. *cygnorum*
Rulingia grandiflora Endl.
Rulingia parviflora Endl.
Thomasia brachystachys Turcz.
Thomasia cognata Steud.
Thomasia discolor Steud.
Thomasia foliosa Gay
Thomasia grandiflora Lindley
Thomasia laxiflora Benth.
Thomasia macrocalyx Steud.
Thomasia pauciflora Lindley
Thomasia multiflora E. Pritzel
Thomasia purpurea (Aiton) Gay
(E) *Thomasia quercifolia* (Andrews) Gay
Thomasia rhynchocarpa Turcz.
(E) *Thomasia solanacea* Gay
Thomasia triloba Turcz.
Thomasia triphylla Gay

DILLENIACEAE

Hibbertia acerosa (R. Br. ex DC.) Benth.
Hibbertia amplexicaulis Steud.
Hibbertia commutata Steud.
Hibbertia cuneiformis (Labill.) Sm.
Hibbertia cunninghamii Ait. ex Hook
Hibbertia furfuracea (R. Br. ex DC.) Benth.
Hibbertia glaberrima F. Muell.
Hibbertia glomerata Benth.
Hibbertia grossulariifolia (Salisb.) Salisb.
Hibbertia hypericoides (DC.) Benth.
Hibbertia inconspicua Ostenf.
Hibbertia lasiopus Benth.
Hibbertia microphylla Steud.
Hibbertia pachyrrhiza Steudel
Hibbertia perfoliata Endl.
Hibbertia pulchra Ostenf.
Hibbertia quadricolor Domin
Hibbertia racemosa (Endl.) Gilg
Hibbertia rhadinopoda F. Muell.
Hibbertia serrata Hotchk.
Hibbertia silvestris Diels
Hibbertia stellaris Endl.
Hibbertia subvaginata (Benth.) F. Muell.

CLUSIACEAE

Hypericum gramineum G. Forster
Hypericum japonicum Thunb.
* *Hypericum perforatum* L.

FRANKENIACEAE

Frankenia pauciflora DC.
Frankenia tetrapetala Labill.

VIOLACEAE

Hybanthus calycinus (DC. ex Ging.) F. Muell.

Hybanthus debilissimus F. Muell.
Hybanthus floribundus (Lindley) F. Muell. ssp. *floribundus*
Hybanthus volubilis E.M. Bennett
* *Viola odorata* L.

THYMELAEACEAE
Pimelea angustifolia R. Br.
Pimelea argentea R. Br.
Pimelea brevifolia R. Br. ssp. *brevifolia*
Pimelea ciliata B.L. Rye ssp. *ciliata*
Pimelea clavata Labill.
Pimelea cracens B.L. Rye ssp. cracens
Pimelea cracens ssp. *glabra* B.L. Rye
Pimelea ferruginea Labill.
Pimelea hispida R. Br.
Pimelea imbricata R. Br. ssp. *imbricata*
Pimelea imbricata ssp. *piligera* B.L. Rye
Pimelea lanata R. Br.
Pimelea lehmanniana Meissner ssp. *lehmanniana*
Pimelea longiflora R. Br. ssp. *longiflora*
Pimelea preissii Meissner
Pimelea rosea R. Br.
Pimelea spectabilis Lindley
Pimelea suaveolens Meissner ssp. *suaveolens*
Pimelea sylvestris R. Br.
Pimelea tinctoria Meissner

LYTHRACEAE
* *Lythrum hyssopifolia* L.

MYRTACEAE
Actinodium cunninghamii Schauer
Agonis flexuosa (Sprengel) Schauer
Agonis floribunda Turcz.
Agonis hypericifolia Schauer
Agonis juniperina Schauer
Agonis linearifolia (DC.) Schauer
Agonis marginata (Labill.) Schauer
Agonis parviceps Schauer
Agonis undulata Benth.
Astartea clavulata Turcz.
Astartea aff. *fasicularis* (Labill.) DC.
(E) *Astartea* sp. nov (GK 970)
Baeckea arbuscula R. Br. ex Benth.
Baeckea astarteoides Benth.
Baeckea blacketii F. Muell.
Baeckea camphorosmae Endl.
Baeckea pygmaea R. Br. ex Benth.
Beaufortia anisandra Schauer
Beaufortia decussata R. Br.
Beaufortia micrantha Schauer
Beaufortia sparsa R. Br.
Beaufortia squarrosa Schauer
Callistemon glaucus Bonpl. Sweet
Calothamnus gracilis R. Br.
Calothamnus graniticus T.J. Hawkeswood ssp. *graniticus*
Calothamnus lateralis Lindley
Calothamnus preissii Schauer
Calothamnus sanguineus Labill.
Calothamnus schaueri Lehm.
Calytrix acutifolia (Lindley) Craven
Calytrix asperula (Schau.) Benth.
Calytrix birdii (F. Muell.) B.D. Jackson
Calytrix flavescens Cunn.
Calytrix leschenaultii (Schauer) Benth.
Calytrix tenuiramea (Turcz.) Benth.
Calytrix tetragona Labill.
Chamelaucium ciliatum Desf.
(E) *Chamelaucium* sp.
Darwinia citriodora (Endl.) Benth.
Darwinia diosmoides (DC.) Benth.
Darwinia forrestii F. Muell.
Darwinia oederoides (Turcz.) Benth.
Darwinia vestita (Endl.) Benth.
Eremaea pauciflora (Endl.) Druce
Eucalyptus angulosa Schauer
(E) *Eucalyptus brevistylis* Brooker
Eucalyptus calophylla Lindley
(E) *Eucalyptus calcicola* Brooker
Eucalyptus conferruminata D. Carr et S. Carr
Eucalyptus cornuta Labill.
Eucalyptus decipiens Endl.
Eucalyptus decurva F. Muell.
Eucalyptus diversicolor F. Muell.
Eucalyptus doratoxylon F. Muell.
Eucalyptus drummondii Benth.
Eucalyptus falcata Turcz.
(E) *Eucalyptus ficifolia* F. Muell.
Eucalyptus goniantha Turcz. ssp. nov.
(E) *Eucalyptus guilfoylei* Maiden
Eucalyptus jacksonii Maiden
Eucalyptus loxophleba Benth.
Eucalyptus marginata Donn. ex Smith
Eucalyptus megacarpa F. Muell.
Eucalyptus missilis Brooker & Hopper ined.
* *Eucalyptus muelleriana*
Eucalyptus patens Benth.
Eucalyptus rudis Endl. ssp. *rudis*
Eucalyptus rudis Endl. ssp. *cratyantha* Brooker & Hopper ined.
Eucalyptus staeri (Maiden) Kessell et C. Gardner
Eucalyptus wandoo Blakely
Homalospermum firmum Schauer
Hypocalymma angustifolium Endl.
Hypocalymma cordifolium (Lehm.) Schauer
Hypocalymma ericifolium Benth.
Hypocalymma robustum Endl.
Hypocalymma strictum Schauer
(E) *Hypocalymma* sp.
Kunzea ericifolia (Smith) Heynh.
Kunzea aff. *micrantha* Schauer

Kunzea spicata S. Moore
Kunzea recurva Schauer var. *recurva*
Kunzea recurva var.? *melaleucoides*
(E) *Kunzea sulphurea* Tovey et Morris
Kunzea vestita Schauer
Leptospermum erubescens Schauer
* *Leptospermum laevigatum* (Gaertner) F. Muell.
Melaleuca acerosa Schauer
(E) *Melaleuca basicephala* Benth.
Melaleuca baxteri Benth.
Melaleuca bracteosa Turcz.
Melaleuca cuticularis Labill.
Melaleuca densa R. Br.
Melaleuca diosmifolia Andrews
Melaleuca huegelii Endl.
Melaleuca incana R. Br.
Melaleuca lanceolata Otto
Melaleuca laterita Otto L.A. Dietr.
Melaleuca leptoclada Benth.
Melaleuca micromera Shauer
Melaleuca microphylla Smith
Melaleuca pauciflora Turcz.
Melaleuca pentagona Labill.
Melaleuca polygaloides Schauer
Melaleuca preissiana Schauer
Melaleuca rhaphiophylla Schauer
Melaleuca scabra R. Br. var. *trichophylla*
Melaleuca ?seriata Lindley
Melaleuca spathulata Schauer
Melaleuca striata Labill.
Melaleuca thymoides Turcz.
Melaleuca viminea Lindley
Melaleuca violacea Lindley
(E) *Pericalymma crassipes* (Endl.) Schauer
Pericalymma ellipticum (Endl.) Schauer
Scholtzia sp.
Thryptomene saxicola (Cunn. ex Hook.) Schauer
Thryptomene aff. *hyporhytis* Turcz.
Verticordia acerosa Lindley
Verticordia densiflora Lindley
Verticordia habrantha Schauer
Verticordia lehmannii Schauer
Verticordia lindleyi Schauer
Verticordia pennigera Endl.
Verticordia plumosa (Desf.) Druce

ONAGRACEAE

Epilobium billardierum Ser. ssp. *billardierum*
Epilobium billardierum ssp. *cinereum* (A. Rich.) Raven et Englehorn
Epilobium billardierum ssp. *intermedium* Raven et Englehorn
* *Epilobium ciliatum* Raf.
Epilobium hirtigerum Cunn.
* *Oenothera glazioviana* Micheli
* *Oenothera stricta* Ledeb. ex Link.

HALORAGACEAE

Glischrocaryon aureum (Lindley) Orch. var. *aureum*
Glischrocaryon aureum var. *angustifolium* (Nees) Orch.
Glischrocaryon roei Endl.
Gonocarpus benthamii Orch.
Gonocarpus diffusus (Diels.) Orch.
(E) *Gonocarpus hexandrus* (F. Muell.) Orch. ssp. *hexandrus*
(E) *Gonocarpus hexandrus* ssp. *serratus* (Schindl.) Orch.
Gonocarpus nodulosus Nees
Gonocarpus panniculatus (R. Br. ex Benth.) Orch.
Gonocarpus simplex (R. Br. ex Britt.) Orch.
Haloragis acutangula F. Muell. forma *occidentalis* Orch.
Haloragis brownii (J.D. Hook) Schindler
Haloragis ?digyna Labill.
Haloragodedron racemosum (Labill.) Orch.
(E) *Meziella trifida* (Nees) Schindler
* *Myriophyllum aquaticum* (Vell. Conc.) Verdc.
Myriophyllum crispatum Orch.
Myriophyllum drummondii Benth.
Myriophyllum salsugineum Orch.

APIACEAE

(E) *Actinotus 'laxa'*
Actinotus omnifertilis F. Muell. ex Benth.
Actinotus glomeratus Benth.
* *Ammi majus* L.
Apium annuum P.S. Short
Apium prostratum Labill. ex Vent. ssp. *prostratum*
Apium prostratum var. *filiforme* (A. Rich.) Kirk
* *Centella asiatica* (L.) Urban
* *Conium maculatum* L.
* *Daucus carota* L.
Daucus glochidiatus (Labill.) Fischer
Eryngium pinnatifidum Bunge
* *Foeniculum vulgare* Miller
Homalosciadium homalocarpum (F. Muell.) Hj. Eichler
Hydrocotyle alata R. Br.
Hydrocotyle blepharocarpa F. Muell.
Hydrocotyle callicarpa Bunge
Hydrocotyle diantha DC.
Hydrocotyle hirta R. Br. ex A. Rich.
Hydrocotyle hispidula Bunge. var. *hispidula*
(E) *Hydrocotyle hispidula* var. *tenella* Benth.
Hydrocotyle medicaginoides Turcz.
Hydrocotyle pilifera Turcz. var. *glabrata* Benth.
Hydrocotyle plebeja R. Br. ex A. Rich.
Hydrocotyle scutellifera Benth.
Hydrocotyle tetragonocarpa F. Muell.
Hydrocotyle sp. (Hamelin Bay)
Pentapeltis peltigera (Hook) Bunge
Pentapeltis silvatica (Diels) Domin
Platysace anceps (DC.) Norman

Platysace compressa (Labill.) Norman
Platysace filiformis (Bunge.) Norman
Platysace haplosciadia (Benth.) Norman
Platysace ramosissima (Benth.) Norman
Platysace pendula (Benth.) Norman
Platysace tenuissima (Benth.) Norman
Schoenolaena juncea Bunge.
Schoenolaena tenuior Bunge.
Sium latifolium L.
Trachymene anisocarpa (Turcz.) B.L. Burtt
Trachymene coerulea R.A. Graham
Trachymene ornata (Endl.) Druce
Trachymene pilosa Sm.
Xanthosia atkinsoniana F. Muell.
Xanthosia candida (Benth.) Steudel
Xanthosia hederifolia Benth.
Xanthosia huegelii (Benth.) Steudel
Xanthosia pusilla Bunge.
Xanthosia rotundifolia DC.
(?E)Gen Nov/Sp. Nov. (Shannon)

EPACRIDACEAE

Actrotriche cordata (Labill.) R. Br.
Actrotriche depressa R. Br.
(E) *Andersonia auriculata* L. Watson
(E) *Andersonia barbata* L. Watson
Andersonia caerulea R. Br.
Andersonia involucrata Sonder
Andersonia lehmanniana Sonder ssp. *lehmanniana*
Andersonia longifolia (Benth.) L. Watson
Andersonia micrantha R. Br.
Andersonia simplex (Stschegl.) Druce
Andersonia sprengelioides R. Br.
Andersonia sp. I (Annels 4064)
Andersonia sp. II (Hamersley 335)
Astroloma baxteri DC.
Astroloma ciliatum (Lindley) Druce
Astroloma drummondii Sonder
Astroloma epacridis (DC.) Druce
Astroloma humifusum (Cav.) R. Br.
Astroloma pallidum R. Br.
Astroloma prostratum R. Br.
Brachyloma concolor (F. Muell.) C. Gardner
Brachyloma preissii Sonder
Conostephium preissii Sonder
Cosmelia rubra R. Br.
(E) *Leucopogon alternifolius* R. Br.
Leucopogon assimilis R. Br.
Leucopogon australis R. Br.
Leucopogon bracteolaris Benth.
Leucopogon capitellatus DC.
Leucopogon carinatus
Leucopogon cinereus E. Pritzel
Leucopogon concinnus Benth.
Leucopogon conostephioides DC.
Leucopogon cordatus Sonder
Leucopogon cucullatus R. Br.
(E) *Leucopogon denticulatus* W.V. Fitzg.
Leucopogon distans R. Br.
Leucopogon elatior Sonder
Leucopogon flavescens Sonder
(E) *Leucopogon gilbertii* Stschegl.
(E) *Leucopogon* aff. *gilbertii* (CJR 192)
Leucopogon glabellus R. Br.
Leucopogon gracilis R. Br.
Leucopogon gracillimus DC.
Leucopogon hirsutus Sonder
Leucopogon kingianus (F. Muell.) C. Gardner
Leucopogon multiflorus R. Br.
Leucopogon nutans E. Pritzel
Leucopogon obovatus (Labill.) R. Br.
Leucopogon oppositifolius Sonder
Leucopogon ovalifolius Sonder
Leucopogon oxycedrus Sonder
Leucopogon parviflorus (Andr.) Lindley
Leucopogon pendulus R. Br.
Leucopogon polymorphus Sonder
(E) *Leucopogon polystachyus* R. Br.
Leucopogon propinquus R. Br.
Leucopogon racemulosus DC.
Leucopogon reflexus R. Br.
Leucopogon sprengelioides Sonder
Leucopogon striatus R. Br.
Leucopogon strictus Benth.
Leucopogon aff. *tenuis* DC.
Leucopogon unilateralis Stschegl.
Leucopogon verticillatus R. Br.
Lysinema ciliatum R. Br.
Lysinema sp. aff. *ciliatum*
Lysinema conspicuum R. Br.
Lysinema fimbriatum F. Muell.
Lysinema lasianthum R. Br.
Monotoca tamariscina F. Muell.
Needhamiella pumilio (R. Br.) L. Watson
Oligarrhena micrantha R. Br.
Sphenotoma capitatum (R. Br.) Lindley
Sphenotoma gracile (R. Br.) Sweet
Sphenotoma parviflorum F. Muell.
Sphenotoma squarrosum (R. Br.) Don
Styphelia tenuiflora Lindl.

PRIMULACEAE

* *Anagallis arvensis* L. var. *arvensis*
* *Anagallis arvensis* var. *caerulea* Gouan
Samolus junceus R. Br.
Samolus repens (Forster et G. Forster) Pers.
Samolus valerandi L.

LOGANIACEAE

Logania buxifolia F. Muell.
Logania campanulata R. Br.
Logania fasciculata R. Br.
Logania serpyllifolia R. Br.
Logania vaginalis (Labill.) F. Muell.

Logania sp. aff. *serphyllifolia* (GK 10371)
Mitrasacme paradoxa R. Br.
Mitrasacme sp. (Annels 2706)

GENTIANACEAE
* *Centaurium erythraea* Rafn.
* *Centaurium spicatum* (L.) Fritsch
Sebaea ovata (Labill.) R. Br.

MENYANTHACEAE
Villarsia albiflora F. Muell.
Villarsia capitata Nees.
Villarsia lasiosperma F. Muell.
Villarsia latifolia Benth.
Villarsia parnassiifolia (Labill.) R. Br.
Villarsia submersa Aston
Villarsia violifolia F. Muell.

APOCYNACEAE
* *Vinca major* L.

ASCELPIADACEAE
* *Gomphocarpus fruticosus* (L.) W.T. Aiton

CONVOLVULACEAE
?* *Calystegia soldanella* R. Br.
Dichondra repens Forster et G. Forster
* *Ipomaea indica* (Burrman) Merr.
Wilsonia backhousii J.D. Hook.
Wilsonia humilis R. Br.

CUSCUTACEAE
Cuscuta australis R. Br.

BORAGINACEAE
* *Borago officinalis* L.
* *Echium plantagineum* L.
Myosotis australis R. Br.

VERBENACEAE
* *Verbena bonariensis* L.

CHLOANTHACEAE
Pityrodia bartlingii (Lehm.) Benth.

LAMIACEAE
Hemiandra pungens R. Br. var. *pungens*
Hemigenia incana (Lindley) Benth.
Hemigenia microphylla Benth.
Hemigenia podalyrina F. Muell.
Hemigenia sericea Benth.
* *Mentha aquatica* L.
* *Mentha* x *piperita* L.
* *Mentha pulegium* L.
* *Mentha spicata* L.
* *Mentha suaveolens* Ehrh.
Microcorys aff. *obvata* Benth.
* *Prunella vulgaris* L.
* *Salvia verbenacea* L.
Westringia dampieri R. Br.

SOLANACEAE
Anthocercis littorea Labill.
Anthocercis viscosa R. Br. ssp. *viscosa*
(E) *Anthocercis* sp. (Annels 4036)
* *Datura stramonium* L.
* *Lycium ferocissimum* Miers
* *Nicandra physalodes* (L.) P. Gaertner
* *Physalis peruviana* L.
* *Solanum laciniatum* Aiton
* *Solanum nigrum* L.
Solanum symonii Hj. Eichler

SCROPHULARIACEAE
* *Bellardia trixago* (L.) All.
* *Dischisma arenarium* E. Meyer
Euphrasia collina R. Br. ssp. *tetragona* (R. Br.) W.R. Barker
Euphrasia scabra R. Br.
Glossostigma drummondii Benth.
Grattiola peruviana L.
Morgania floribunda? Benth.
* *Parentucellia latifolia* (L.) Caruel
* *Parentucellia viscosa* (L.) Caruel
* *Verbascum virgatum* Stokes
* *Veronica arvensis* L.
Veronica calycina R. Br.
Veronica distans R. Br.
Veronica plebeia R. Br.

OROBANCHACEAE
* *Orobanche minor* Smith

LENTIBULARIACEAE
Polypomphylx multifida (R. Br.) F. Muell.
Polypomphlyx tenella (R. Br.) Lehm.
Utricularia menziesii R. Br.
Utricularia multifida R. Br.
Utricularia tenella R. Br.
Utricularia simplex R. Br.
Utricularia violacea R. Br.
Utricularia volubilis R. Br.

MYOPORACEAE
Myoporum apiculatum A. DC.
Myoporum gracile Bartling
Myoporum insulare R. Br.
Myoporum oppositifolium R. Br.
Myoporum tetrandrum (Labill.) Domin

PLANTAGINACEAE
Plantago debilis R. Br.
* *Plantago lanceolata* L.
* *Plantago major* L.

RUBIACEAE

* *Galium murale* (L.) All.
Opercularia echinocephala Benth.
Opercularia hispidula Endl.
Opercularia vaginata Labill.
Opercularia volubilis R. Br. ex Benth.

VALERIANACEAE

* *Centranthus ruber* (L.) DC.

CUCURBITACEAE

* *Cucumis myriocarpus* Naudin

CAMPANULACEAE

Wahlenbergia communis Carolin
Wahlenbergia gracilenta Lothian
Wahlenbergia graniticola Carolin
Wahlenbergia litticola P.J. Smith
Wahlenbergia multicaulis Benth.
Wahlenbergia preissii Vriese
Wahlenbergia simplicicaulis Vriese

LOBELIACEAE

Grammatotheca bergiana (Cham.) C. Presl
Isotoma hypocratiformis (R. Br.) Druce
Isotoma scapigera (R. Br.) G. Don.
Lobelia alata Labill.
Lobelia gibbosa Labill.
Lobelia heterophylla Labill.
Lobelia rhombifolia Vriese
Lobelia rhytidosperma Benth.
Lobelia rariflora F. Wimmer
Lobelia tenuior R. Br.
* *Monopsis simplex* (L.) E. Wimm.

GOODENIACEAE

Anthotium junciforme (De.Vr.) Morrison
Dampiera fasiculata R. Br.
Dampiera hederacea R. Br.
(E) *Dampiera heteroptera* Rajput et Carolin
Dampiera leptoclada Benth.
Dampiera linearis R. Br.
Dampiera pedunculata Rajput et Carolin
Dampiera trigona De Vriese
Diaspasis filifolia R. Br.
Goodenia caerulea R. Br.
Goodenia concinna Benth.
Goodenia eatoniana F. Muell.
Goodenia filiformis R. Br. var. *filiformis*
Goodenia filiformis var. *pulchella* Benth.
Goodenia incana R. Br.
Goodenia laytoniana Benth.
Goodenia leptoclada Benth.
Goodenia pulchella Benth.
Goodenia sepalosa F. Muell. ex Benth. var. *glandulosa* F. Muell.
Goodenia tenella R. Br.
Lechenaultia biloba Lindley
Lechenaultia expansa R. Br.
Lechenaultia floribunda Benth.
Lechenaultia formosa R. Br.
Lechenaultia tubiflora R. Br.
(E) *Scaevola attenuata* R. Br.
Scaevola auriculata Benth.
Scaevola calliptera Benth.
Scaevola crassifolia Labill.
Scaevola glandulifera DC.
Scaevola globulifera Labill.
Scaevola lanceolata Benth.
Scaevola longifolia Vriese
Scaevola microphylla Benth.
Scaevola nitida R. Br.
Scaevola pilosa Benth.
Scaevola striata R. Br.
Scaevola thesioides Benth.
Selliera radicans Cav.
(E) *Velleia macrophylla* (Lindley) Benth.
Velleia trinervis Labill.

STYLIDIACEAE

Levenhookia dubia Sonder
Levenhookia leptantha Benth.
Levenhookia pauciflora Benth.
Levenhookia preissii (Sonder) F. Muell.
Levenhookia pusilla R. Br.
Stylidium adnatum R. Br.
Stylidium affine Sonder
Stylidium amoenum R. Br.
Stylidium assimile R. Br.
Stylidium barleei F. Muell.
Stylidium beaugleholei J.H. Willis
Stylidium breviscapum R. Br.
Stylidium brunonianum Benth. ssp. *brunonianum*
Stylidium brunonianum Benth. ssp. *minor* Carlq.
Stylidium bulbiferum Benth.
Stylidium caespitosum R. Br.
Stylidium calcaratum R. Br.
Stylidium aff. *calcaratum* R. Br.
Stylidium canaliculatum Lindley
Stylidium carnosum Benth.
Stylidium ciliatum Lindley
Stylidium corymbosum R. Br.
Stylidium crassifolium R. Br.
Stylidium despectum R. Br.
Stylidium dichotomum DC.
Stylidium ecorne (F. Muell. ex R. Erickson et J.H. Willis) P.G. Farrell et S.H. James
Stylidium exoglossum R. Erickson et J.H. Willis
Stylidium falcatum R. Br.
Stylidium fasciculatum R. Br.
Stylidium glaucum Labill. ssp. *glaucum*
(E) *Stylidium glaucum* ssp. *angustifolium* Carlq.
Stylidium guttatum R. Br.
Stylidium hirsutum R. Br.
Stylidium imbricatum Benth.

Stylidium inundatum R. Br.
Stylidium junceum R. Br. ssp. *junceum*
Stylidium junceum ssp. *brevis* (E. Pritzel.) Carlq.
(E) *Stylidium laciniatum* C. Gardner
Stylidium lepidium F. Muell. ex Benth.
Stylidium luteum R. Br. ssp. *luteum*
Stylidium luteum ssp, *glaucifolium* Carlq.
Stylidium periscelianthum R. Erickson et J.H. Willis
Stylidium perpusillum J.D. Hook
Stylidium petiolare Sonder
Stylidium piliferum R. Br.
Stylidium preissii (Sonder) F. Muell.
(E) *Stylidium pritzelianum* Milbr.
Stylidium pulchellum Sonder
(E) *Stylidium pygmaeum* R. Br.
Stylidium rhynchocarpum Sonder
Stylidium repens R. Br.
Stylidium rupestre Sonder
Stylidium scandens R. Br.
(E) *Stylidium* aff. *scandens* R. Br.
Stylidium schoenoides DC.
Stylidium spathulatum R. Br. ssp. *spathulatum*
(?E)*Stylidium spathulatum* ssp. *acuminatum* Carlq.
Stylidium spinulosum R. Br.
Stylidium squamosotuberosum Carlq.
Stylidium uniflorum Sond.
Stylidium violaceum R. Br.

ASTERACEAE
Actites megalocarpa (J.D. Hook) N.S. Lander
Angianthus preissianus (Steetz.) Benth.
* *Arctotheca calendula* (L.) Levyns
* *Arctotheca populifolia* (P. Bergius) Norlindh
Asteridea gracilis A. Gray
Asteridea nivea (Steetz.) G. Kroner
Asteridea pulverulenta Lindley
Berkheya rigida (Thunb.) Ewart, J. White et B. Rees
Blennospora drummondii A. Gray
Brachycome ciliaris (Labill.) Less
Brachycome exilis Sonder
Brachycome iberidifolia Benth.
Calocephalus brownii (A. Gray) Benth.
* *Carduus pycnocephalus* L.
* *Carduus tenuiflorus* Curt
* *Centaurea melitensis* L.
Centipeda cunninghamii (DC.) A. Braun et Asch.
* *Chrysanthemum segetum* L.
Chrysocoryne pusilla (Benth.) Endl.
* *Cirsium arvense* (L.) Scop.
* *Cirsium vulgare* (Savi) Ten.
* *Conyza albida* Willd. ex Spreng.
* *Conyza bonariensis* (L.) Cronq.
* *Conyza parva* Cronq.
* *Coreopsis grandiflora* Hogg ex Sweet
Cotula australis (Sieber ex Sprengel) J.D. Hook.
Cotula coronopifolia L.
Cotula cotuloides (Steetz) Druce
Cotula drummondii Benth.
* *Cotula turbinata* L.
Craspedia pleiocephala F. Muell.
* *Crepis foetida* L.
* *Cynara cardunculus* L.
* *Dittrichia graveolens* (L.) Greuter
* *Dittrichia viscosa* (L.) Greuter
Gnaphalium gymnocephalum DC.
Gnaphalium indutum J.D. Hook.
* *Gnaphalium pensylvanicum* Willd.
Gnaphalium sphaericum Willd.
* *Hedyponis rhagadioloides* (L.) F.W. Schmidt
Helichrysum cordatum DC.
Helichrysum macranthum Benth.
Helichrysum obtusifolium F. Muell. et Sonder
Helichrysum ramosum DC.
Helipterum pygmaeum (DC.) Benth.
Hyalosperma cotula (Benth.) P.G. Wilson
Hyalosperma pusillum (Turcz.) P.G. Wilson
Hyalosperma simplex (Steetz.) P.G. Wilson
* *Hypochaeris glabra* L.
Ixiolaena viscosa Benth.
Lagenifera huegelii Benth.
Leptorhynchos nudius
Leptothynchos scabrus L. Haegi
Millotia myosotidifolia (Benth.) Steetz
Millotia tenuifolia Cass
Olearia axillaris (DC.) F. Muell. et Benth.
Olearia calcarea F. Muell.
Olearia cassiniae (F. Muell.) Benth.
Olearia ciliata (Benth.) F. Muell.
Olearia elaeophila (DC.) F. Muell. ex Benth.
Olearia paucidentata (Steetz) F. Muell. ex Benth.
Olearia revoluta F. Muell. ex Benth.
Olearia rudis (Benth.) F. Muell. ex Benth.
* *Osteospermum clandestinum* (Lees.) Norlindh.
* *Pentzia suffruticosa* (L.) Druce
Picris squarrosa Steetz
Pithocarpa corymbulosa Lindley
Pithocarpa melanostigma Lewis et Summerh.
Podolepis canescens A. Cunn. ex DC.
Podolepis gracilis (Lehm.) R. Graham
Podolepis lessonii (Cass.) Benth.
Podolepis rugata Labill.
Podotheca angustifolia (Labill.) Less.
* *Pseudognaphalium luteo-album* (L.) Hilliard et B.L. Burtt
Quinetia urvillei Cass.
Rutidosis multiflora (Nees) Robinson
* *Senecio diaschides* Drury
* *Senecio elegans* L.
Senecio glomeratus Desf. ex Poiret
Senecio glossanthus (Sonder) Belcher
Senecio hispidulus
* *Senecio jacobaea* L.
Senecio lautus G. Forster ex Willd. ssp. *maritimus* Ali
* *Senecio mikanioides* Otto ex Wal.

Senecio minimus Poiret var. *minimus*
Senecio minimus Poiret var. *picridiodes* Benth.
Senecio quadridentatus Labill.
Senecio ramosissimus DC.
* *Senecio vulgaris* L.
* *Siegsbeckia orientalis* L.
Siloxerus filifolius (Benth.) Ostenf.
Siloxerus humifusus Labill.
* *Silybum marianum* (L.) Gaertner
* *Soliva pterosperma* (A.L. Juss.) Less.
* *Sonchus asper* Hill
* *Sonchus oleraceus* L.
Trichocline spathulata (A. Cunn. ex DC.) J.H. Willis
* *Ursinia anthemoides* (L.) Poiret
* *Ursinia speciosa* DC.
* *Vellereophyton dealbatum* (Thunb.) Hilliard et B.L. Burtt.
Waitzia citrina Steetz
Waitzia paniculata (Steetz) Benth.
Waitzia suaveolens (Benth.) Druce
* *Xanthium spinosum* L.

CHAPTER 2

A Review of the Effects of Disturbance on Wildlife of the Karri Forest

Grant Wardell-Johnson and Per Christensen

Abstract

This paper reviews current knowledge of ongoing research, and future research priorities, into the effects of disturbance on the flora and fauna of the karri forest and other communities in the Warren Botanical Subdistrict. Research on harvesting and regeneration, fire, naturalized species, and relevant biogeographic and evolutionary studies are considered in areas of retained native vegetation.

Biogeographic and evolutionary studies indicate that past climatic changes have led to changes in the distribution of species. Subsequent activities by humans, however, have been of greater significance in the decline of vulnerable species than have climatic changes. The development of an understanding of the overall pattern of the biota and the distribution of vulnerable species (including Gondwanan relicts) and their habitats remain a priority in biogeographic research.

Studies on harvesting and regeneration have maintained emphasis on silvicultural techniques to optimize timber production from regenerated stands. Wildlife research on the effects of timber harvesting has concentrated on birds, contributing to recommendations for a redistribution of road, river and stream zones to more effectively cater for fauna conservation in multiple use forests. The process of hollow formation in trees, the maintenance of vulnerable species, and the predictive modelling of community changes are research priorities in relation to harvesting and regeneration.

Studies on the effects of fire have been wide ranging including work on vascular plants, small mammals, birds and invertebrates, though gaps remain in studies that would allow predictive modelling and an understanding of fire responses of invertebrate communities and of vulnerable plant species. Continued research on the control of key naturalized species (e.g. *Phytophthora* and Red fox) and on the maintenance of community processes in the presence of these species are also priorities.

A multidisciplinary approach to process oriented research is recommended in disturbance ecology. Regional staff take an increasing role in ecological monitoring and experimental management.

INTRODUCTION

Community types containing karri (*Eucalyptus diversicolor*) occur in the Warren and Menzies Subdistricts of the Darling Botanical District (Beard 1980) in south-western Australia. The principal occurrence of karri is in the Warren Subdistrict (Fig. 1) with outliers in the Porongurup Range and near Manypeaks. Most land (169 000 ha) that now includes karri is administered by the Department of Conservation and Land Management (CALM) as State forests, nature reserves, national parks and conservation parks.

Karri occurs in many community types and the effects of disturbance may extend across community types. Hence we include the Warren Botanical Subdistrict as our area of coverage because it includes all karri State forest as well as neighbouring community types and enables placement of those components confined to the karri forest in a regional perspective. The boundaries of the Warren Subdistrict are defined by Beard (1980) and outlined in detail by Hopper *et al.* (this volume).

Disturbance can be natural, or owing to human interference. Humans have blurred the boundaries of natural disturbance, and virtually all forms of disturbance are now influenced by human activities. As Havel (1989) and Mills (1989) demonstrate, human impacts, particularly in the south-west forests, have increased greatly since European settlement, particularly when land use intensified in the 1960s. There have been major changes in the patterns of disturbance, few of which can be attributed to individual factors. Hobbs and Hopkins (1990) recognized four main categories of land use and resultant degrees of impact following European modification of Australian vegetation. These include:

(1) complete removal of the vegetation and disruption of ecological processes such as mining and urban development;

(2) replacement of vegetation by intensively managed systems such as agriculture and plantation forestry;

(3) utilization of existing vegetation with some consequent modification as occurs in forests managed for multiple uses; and

(4) management for nature conservation with minimal deliberate modification.

Concern has been expressed about the adequacy of knowledge of the long term effects of disturbance on the composition of the flora and fauna of the karri forest (Anon. 1986; Christensen 1986; Anon. 1988). Here we review the current state of knowledge and the existing research program on the effects of disturbance on the wildlife of the karri forest. We consider only the third and fourth of the categories of land use recognized by Hobbs and Hopkins (1990). Disturbances included within these categories include fire, logging and regeneration, and the effects of introduced species.

The long term effects of disturbance on the composition of the flora and fauna will be intimately tied to interactions with climatic change and with effects on soils and nutrients. These subjects are reviewed by Bartle (this volume), Shearer (this volume) and Abbott (this volume), and by Christensen and Abbott (1989) and we do not intend to address these issues here. Aquatic ecosystems are also reviewed elsewhere (Halse and Blyth, this volume). We therefore confine this review to the terrestrial biota, but take a wide view of long term effects. We also provide recommendations for future research.

CURRENT KNOWLEDGE

Evolutionary and biogeographical studies

FLORA

Research on the effects of disturbance on the flora of the Warren Subdistrict has been concentrated in karri forest in the Nornalup System. Inions *et al.* (1990) provided an overview of community types in which karri is a component. Site-based community studies provide benchmarks for monitoring change in the community types investigated. This study provided a floristic classification of community types in regrowth karri from 204 quadrats and 105 species. Thirteen community types were defined by cluster analysis, ordination and discriminant analysis of the 312-m^2 quadrats.

Inions *et al.* (1990) found that community types varied substantially in productivity, as measured by age-standardized top height and that this in turn was related to climatic and edaphic factors. Variables relating to rainfall distribution, radiation levels, soil acidity and phosphorus levels were found to be the most discriminatory between community types and each differed in stand productivity and in climatic and edaphic variables. The distribution of the community types defined, is broadly geographically based (Fig. 1), although overlap occurs within a single landform/soils unit (as defined by Churchward *et al.* 1988).

Climatic and edaphic influences on the distribution of karri have also been studied by Churchill (1968). He concluded that the distribution of karri was primarily determined by water availability expressed as rainfall during the wettest and driest months of the year. He suggested that this distribution has fluctuated markedly during the past 5000 years. Examination of charcoal accumulation in cones taken from peat swamps also enabled Churchill (1968) to present a 7000 year fire history of forest areas.

Griffin (1985) provided a floristic survey of Smiths Brook Nature Reserve; a small area near Manjimup dominated by karri forest. This survey defined and mapped five vegetation units including one artificially created by disturbance. The area was found to be relatively undisturbed by Europeans despite a number of tracks and the presence of many introduced species.

FAUNA

Vertebrate surveys of the Warren Subdistrict completed by the Forests Department of Western Australia are summarized by Christensen *et al.* (1985). These authors list 19 surveys completed in the southern forests between 1970 and 1982, 15 of which were in the Warren Botanical Subdistrict. Lists of vertebrates and vascular plants are presented by survey and by major vegetation type. The distribution and status of each vertebrate species is discussed and an analysis of faunal trends within the region, including relationships with adjacent fauna, is presented. Christensen *et al.* (1985) conclude that the vertebrate fauna of the forested south-west occurs in a faunal continuum with a strong north-south, and lesser east-west, influence that appears to be related to

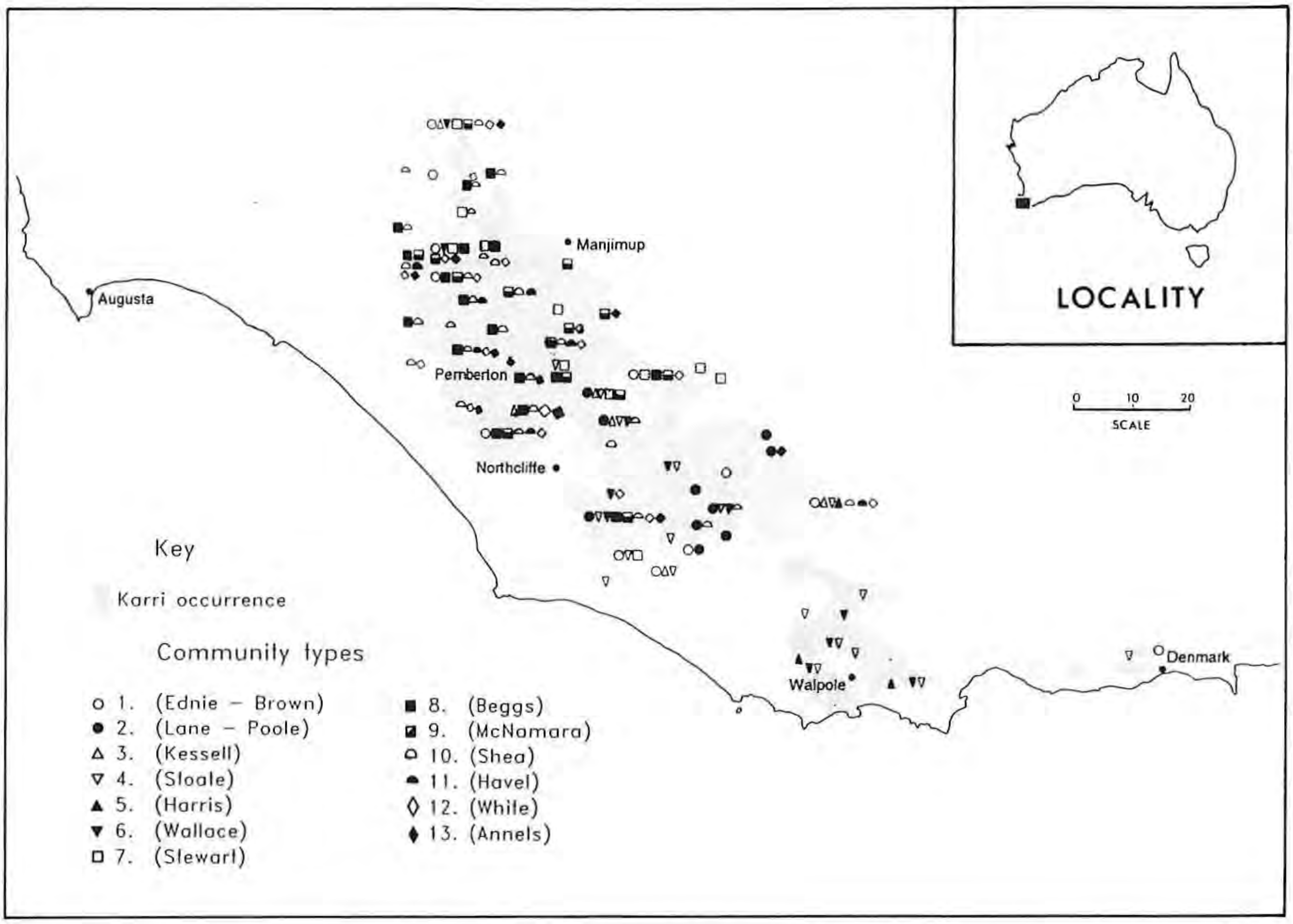

Figure 1
Distribution of community types defined by Inions *et al.* (1990) within the main distribution of karri forest. Major towns are also shown.

climatic factors. On a smaller scale, individual species distribution appeared to be influenced by vegetation and soil factors, although all species occurred across vegetation and soil gradients. Christensen *et al.* (1985) suggested that a possible reason for the lack of diversity of vertebrate fauna in the karri and tingle (*E. guilfoylei* and *E. jacksonii*) forests is that these forests occupy too small an area for dependent residents to have evolved in them.

How *et al.* (1987) present a survey of ground vertebrates and document a decline in numbers of many birds and mammals in the coastal heathlands and woodlands which adjoin the forest between Albany and Busselton, since European settlement. Several vulnerable ground birds and mammals, once present in this coastal area are not now found, or occur in very low numbers; for example, Numbat (*Myrmecobius fasciatus*), Woylie (*Bettongia penicillata*), Tammar (*Macropus eugenii*) and Chuditch (*Dasyurus geoffroyii*). Birds that have become locally extinct following European settlement include the Mallee Fowl (*Leipoo ocellata*), Western Whipbird (*Dasyornis longirostris*), Rufous Bristlebird (*D. broadbenti*), Noisy Scrub-bird (*Atrichornis clamosus*) and Bush Thick-knee (*Burhinus grallarius*).

Similarly, How *et al.* (1987) report a decline to low numbers of the Brush Wallaby (*Macropus irma*), Quokka (*Setonix brachyurus*), Common Brushtail (*Trichosurus vulpecula*) and Western Ringtail (*Pseudocheirus pereginus*) in these south-western areas. This study presents a good historical account of the reasons put forward for these disappearances. For instance, evidence presented by Carter (1923) is cited to explain the disappearance of the Mallee Fowl. According to Carter (1923), the Mallee Fowl was not uncommon in the coastal vegetation between Cape Naturalist and the mouth of the Warren River in 1902. Its populations had diminished by 1920 owing to the burning of the coastal vegetation to improve grazing for cattle. How *et al.* (1987) attributed the decline in ground dwelling vertebrates to a combination of the effects of changed fire regimes associated with the development of pastoralism and clearance for agriculture, introduced predators and competitors (chiefly the Red Fox, *Vulpes vulpes* and Domestic Cat, *Felis catus*) and disease epidemics.

Kabay and Start (1976) reviewed the distributional data on the Broad-faced Potoroo (*Potorous platypus*) and Gilbert's Potoroo (*Potorous tridactylus gilberti*) but found no evidence of the current occurrence of either species in the Warren Botanical Subdistrict. Nevertheless, Gilbert's Potoroo was collected in the vicinity of King George Sound during the 19th century and is well represented as sub-fossil material in cave deposits in Boranup. The habitat requirements of Gilbert's Potoroo were thought to be more restricted than those of the Quokka, itself considerably reduced since European settlement (Kabay and Start 1976). The continued existence of Gilbert's Potoroo in the Warren Botanical Subdistrict while considered unlikely was not discounted. They consider its decline and disappearance may be associated with several factors (disease, predation by foxes and cats, competition, alienation and subsequent clearing of habitat, management practices such as logging, burning and vegetation changes associated with the presence of *Phytophthora cinnamomi*).

Baynes *et al.* (1975) and Balme *et al.* (1978) report on mammal remains spanning about 30 000 years, from excavation in Devil's Lair in Boranup. Both studies suggest that throughout the period studied, a more varied mammal fauna persisted than is recorded in historic time. At some late Holocene period not represented in Devil's Lair, several species known not to inhabit forest disappeared from the district. This and the distribution of mammal species between 35 000 and 5000 years ago suggest that a drier climatic regime in the late Pleistocene gave way to a wetter regime in the Holocene (Balme *et al.* 1978).

Rich and Baird (1986) in presenting fossil evidence of birds from caves in the Boranup area suggest a fluctuating but currently wetter climate than during periods of glaciation. Smith (1977) discussed the effect of environmental change on six rare birds and proposed that all species were more widely distributed and abundant during the Tertiary Period and Pleistocene Epoch - Noisy Scrub-bird, Rufous Scrub-bird (*Atrichornis rufescens*), Rufous Bristlebird, Western Bristlebird, Eastern Bristlebird (*Dasyornis brachysterus*), and Western Whipbird (*Psophodes nigrogularis*) - four of which occurred in the Warren Botanical Subdistrict. Vegetational change from the late Tertiary onwards reduced these species to small remnant populations by the time Europeans arrived. However, the subsequent activities of Europeans (in particular burning and agricultural clearing) are suggested as being of even greater significance to the further decline of these species than that of past climatic changes (Smith 1977, 1985).

Likely effects of climatic change on the distribution of species have not been studied. It is clear that climatic change over aeons has had a marked influence on species distributions and that interaction between disturbance and climatic change may well have important influences in the future.

Harvesting and regeneration

Harvesting for timber in Western Australia began soon after European settlement. Since the passing of the Forests Act in 1919 (Nunn 1957) there has been a continued evolution in forest management and silviculture (Bradshaw and Lush 1981; Abbott and Loneragan 1986; Stoneman 1986; Havel 1989; Stoneman *et al.* 1989; and Breidahl and Hewett 1992).

Clearfelling, which had been replaced by selection cutting in 1938, was reinstated as the main silvicultural system in karri forest in 1967. It was reinstated because it was no longer considered necessary to retain groups of trees as a barrier to alienation of the karri forest, for silvicultural reasons and because of economic efficiencies (Bradshaw and Lush 1981). The woodchip industry was established in 1975 to use non-millable trees from karri forest areas being harvested for sawlogs.

Projected timber demands (Bradshaw and Lush 1981) and supply commitments have led to emphasis being placed on the ability to optimize sawlog production from regenerated stands. Rotheram's (1983) findings that retained trees suppress the growth rate of adjacent regeneration has lent support to this direction. The requirement for thinning of regrowth stands of karri is argued in Bradshaw and Lush (1981). Silvicultural research on karri has been comprehensively reviewed by Breidahl and Hewett (1992) and will not be discussed here other than in the context of disturbance ecology.

Occasional death of karri saplings and understorey species owing to *Armillaria luteobubalina* infection occurs in virgin karri forest and in regrowth karri where it has been related to the presence of nearby infected stumps. There is a decrease in mortality with increasing distance from these stumps (Pearce *et al.* 1986). More suppressed karri saplings were killed than dominant saplings but some degree of natural suppression of *A. luteobubalina* by other fungi has been observed (Pearce and Malajczuk 1990). The distribution and impact of *A. luteobubalina* in jarrah forest has been discussed by Shearer and Tippett (1988) who found that host mortality following infection was greater in the intermediate and low rainfall zones than high rainfall zones of the jarrah forest.

Uncontrolled use of machinery may damage either retained trees in thinning operations (Breidahl and Hewett 1992) or soils, site and tree roots through compaction (Wronski 1984). Breidahl and Hewett (1992) found some evidence of root deformity among planted stock, particularly on badly compacted sites where a hard pan had developed below the surface horizon. Therefore, compaction may affect the establishment and growth of karri and may affect hydrology and thus plant community composition. These problems can be avoided by good management, but in any case are not irreversible (Bradshaw 1978). Schuster (1979a) has examined various methods of rehabilitation of soils disturbed by logging. Deep soil ripping, application of fertilizers and the formation of an ashbed all ameliorated the effects of soil compaction (Schuster 1979b).

FLORISTICS

Inions *et al.* (1990) recorded annually at 144 permanent quadrats all vascular plant species visible in spring in order to compare pre and post logging treatment. Assessment was made before logging or burning on plots that included sites which had not been burnt for 7 to 15 years. Subsequently sites were either burnt (54 sites), clearfelled, burnt and regenerated (50 sites), clearfelled and regenerated (four sites) or left as controls (36 sites). Regeneration was by the seed tree method (18 sites) and by handplanting (36 sites). Assessment continued for five years after the operations and was used to determine whether community type, as defined from the presence or absence of 72 indicator species, remains the same following logging and burning. They concluded that all but one (a quadrat on a log landing) of the 144 quadrats remained as the same community type following these operations. Hence, although their classification was based in regrowth, they concluded that edaphic and climatic variables rather than disturbance were the major determinants of floristic pattern. However, they pointed out through reference to research on other disturbed forest types that many important stand characteristics, such as biomass, species abundance and cover, vary considerably with time since disturbance. Such variation of these characteristics could also be expected in karri forest.

Stoneman *et al.* (1988) measured forest canopy density in a range of different aged stands in karri and jarrah forest and found that total cover in karri forest reached the value of the unlogged stands within five years of regeneration, rose for another five years and then stabilized above the unlogged value. Overstorey canopy cover reached the unlogged value within ten years. They suggested that the rapid recovery in density in karri forest indicates that evapotranspiration is probably close to the pre-logging value within five to ten years of regeneration. Figures 2 to 7 demonstrate the rapid recovery of forest density in karri forest following fire and regeneration.

Breidahl and Hewett (1992) divided their review of silvicultural research into the floral cycle, germination and regeneration; and the management of the regrowth forest. Considerable research has been done on each of these topics.

Loneragan (1979) documents the floral cycle of karri and the influences on the development of the seed crop. Karri seeds periodically (approximately every four years). Production may be expected to be high and regular under high soil moisture conditions and low and irregular under conditions of low soil moisture. The variability of the karri seed cycle poses considerable practical problems in regenerating cutover stands by means of seed trees. Artificial means of regeneration were pioneered in the early 1970s (see Loneragan 1971) and are used for up to 50 per cent of the cutover area in some years (Breidahl and Hewett 1992).

The factors affecting the germination and early survival of karri have been studied in detail. Studies by Meachem (1960), Hatch (1960), Loneragan (1961, 1971), Loneragan and Loneragan (1964), White (1974), and Christensen and Schuster (1979) all concluded that ashbeds are the most favourable sites for the germination and early survival of karri following logging.

There has been relatively less silvicultural research on jarrah (*Eucalyptus marginata*) and marri (*E. calophylla*) in the Warren Subdistrict, although much has been done in the Dale Subdistrict (see Loneragan 1961; Abbott and Loneragan 1986). Much of this work is applicable to the southern jarrah forest and has been used to formulate silvicultural guidelines for the southern jarrah forests (Bradshaw 1986).

The southern jarrah forest has a number of distinctive characteristics including greater variability of soil and landform types (see Churchward *et al.* 1988), more diverse range of understorey (see Strelein 1988) and a generally longer period of moist soil conditions, than the northern jarrah forest. The impacts of dieback disease may be severe in some community types (Grant and Blankendaal 1988; Strelein 1988), although Christensen (1975) reports resistance to the disease in highly productive sites of the southern jarrah forest. Outbreaks of defoliating insects have occurred in the southern jarrah forest. Leaf miner (*Perthida glyphopa*) has been studied by Mazanec (1980) and gum leaf skeletonizer (*Uraba*

Figure 2
Mature forest near Pemberton
(Grant Wardell-Johnson)

Figure 3
A logging coupe in karri forest following a regeneration burn near Pemberton
(Grant Wardell-Johnson)

Figure 4
The same area as in Fig. 3 four years after regeneration
(Grant Wardell-Johnson)

Figure 5
Fourteen-year old regeneration near Pemberton *(Grant Wardell-Johnson)*

Figure 6
Fifty-year old regeneration near Pemberton
(Grant Wardell-Johnson)

Figure 7
Crown in mature karri forest five years afer high intensity wildfire near Pemberton
(Grant Wardell-Johnson)

lugens) by Strelein (1989).The variability of lignotuber development in different sites is also a consideration when planning regeneration following logging operations (Strelein 1988).

Measurements of forest density in jarrah forest by Stoneman *et al.* (1988) show that overstorey canopy cover and total cover, in both high (>1100 mm) and low (<1100 mm) rainfall areas, exceeded 80 per cent of the unlogged value within five years of regeneration. They reached values similar to the unlogged value in about 20 years and thereafter remained at that value. However, forest density recovered more slowly on poorly drained sites than on better drained sites.

INVERTEBRATES

There are few studies on the effects of disturbance on invertebrates in the Warren Botanical Subdistrict. Two unpublished studies by Curry and Humphreys (1987, 1988) used malaise, light and pitfall trapping to sample insect communities in karri forest in spring and summer. Curry and Humphreys (1988) sampled for four years during which the forest was clearfelled, burnt and replanted with karri seedlings. Curry and Humphreys (1987) also sampled pine, jarrah and karri forest. Sampling in karri forest was carried out over six months in four stands of different management history: two-year-old regeneration; two-year-old, not regenerated; 44-year-old regeneration; and unlogged karri forest, seven years since prescription burning.

Curry and Humphreys (1988) discussed the limitations in their sampling design which included a lack of replication and likely site differences that confound treatment effects and the fact that in these studies, sampling was not continued to determine recovery of invertebrate populations with time since regeneration. Nevertheless, they concluded that logging and regeneration resulted in immediate and major changes to the insect community in karri forest. Dissimilarity between ridge and valley sites was found to be large in all sites but was always greater in regeneration than in mature forest. More families were reduced than increased by the disturbance but the response within orders varied widely. The Dipteran families were most affected by the treatment having both the highest proportion of families increased and decreased by the treatment. High sensitivity to disturbance was found for both the Diptera and Lepidoptera suggesting that these groups may be good indicator taxa. The Scolytinae, Leptopiinae, Sphaeroceridae and Nabidae were found only in treated sites and the Lasiocampidae, Arctiidae and Tachinidae in untreated sites, and represent potential indicator groups. At higher resolution some genera were present only on uncleared (*Digglesia, Pterolocera*) or on treatment (*Pollanisus, Calosoma*) sites. Further analysis may provide indicator species for the rapid evaluation of disturbance in karri forest·

Curry *et al.* (1985), in a study of two sites in karri forest over four years, found more aggregated captures and marked changes in the structure of arachnid communities (including changes in species dominance) immediately following clearfelling and regeneration. It has been suggested that relict groups of spiders and others with poor powers of dispersion may not readily re-establish in areas disturbed by logging or fire or by agricultural clearing even if the latter is rehabilitated (Main 1987, 1991).

BIRDS

Tingay and Tingay (1984) reviewed work on the effects of forest operations on bird communities in the karri forest and reported the results of a one-year census in karri forest with a range of management history. Sampling sites consisted of variable aged forest following regeneration after clearfelling and of mature forest which differed in the period since last burnt for fuel reduction purposes. Bird data were collected along fixed transects in each of four seasons, while vegetation was measured with a simple method of habitat description. They demonstrated that clearfelling of karri causes marked changes in vegetation structure, some of which are still evident 50 years afterwards (see Figs 2 to 6). These changes were found to significantly affect bird communities. However, bird species number and abundance increased with forest age. As expected, populations of species common in low, dense vegetation are larger in young regeneration than in mature forest. Many species in mature forest were absent or uncommon in young forest and a few species which are absent or uncommon in older forest occur in recently clearfelled and regenerated sites.

Most species which occur in mature forest were found to be present in sites of 50-year-old regeneration. However, the relative abundance of bird species in the 50-year-old regeneration fluctuated between seasons and was depressed in winter. Those in mature forest were more stable. It was suggested that 50-year-old forest may not have the continuity of niches present in mature forest (Tingay and Tingay 1984).

Tingay and Tingay (1984) recognized the limitations posed by their sampling technique. These limitations are chiefly concerned with likely site differences which may mask effects; and insufficient sampling effort to effectively record relative bird densities. Pike and Recher (1984) recommend that transects be censused over several days to account for differences caused by weather effects. Nevertheless, Tingay and Tingay's (1984) results demonstrated the influence of stand structure on bird community organization earlier demonstrated by Recher (1969, 1971) in a range of vegetation types on different continents. They also demonstrated that bird populations recover in regenerating forests, with time.

Wardell-Johnson (1987) and Christensen (1988) have reviewed the use of hollows by wildlife in the karri forest. Twenty species which use hollows occur in the karri forest and all occur in the Warren Botanical Subdistrict. Wardell-Johnson (1984) found that 14 of 44 species observed in karri forest in spring 1982 used hollows in trees as nest sites including 34 per cent of all bird detections (4327 total). Thus, hollows are important for the total numbers in bird communities in the karri forest as well as the numbers of species.

MAMMALS

Hollows are also important for mammals. Some 67 per cent (20 species) of the species of mammals occurring in the forest of the south-west use hollows. Nine species (30 per cent of the total forest mammal fauna) require hollows in trees in karri forest. However, not all available hollows are used and the study of the use of hollows in karri forest is, in practice, very difficult. Thus, Christensen (1988) found very little evidence of the use of tree hollows by wildlife in karri forest. Wardell-Johnson (1986) demonstrated the pattern of use of nest boxes by Mardos (*Antechinus flavipes leucogaster*) in 14-year-old regenerating karri forest and suggested that hollows may be limiting for this species in regrowth karri forest.

Mammals that use hollows in trees as nest sites have received little study in the Warren Botanical Subdistrict. The interactions between the Common Brushtail and Western Ringtail Possums, habitat trees and fire, however, have been studied in jarrah forest in the Perup Nature Reserve in the adjacent Menzies Subdistrict (Inions 1985; Inions *et al.* 1989).

Fire

Research on fire behaviour was started in the south-west jarrah forest by Peet (1965) and is now well understood in jarrah and karri forest (see Sneeuwjagt 1971; McCaw and Burrows 1989). Predictive models of fire behaviour in jarrah and karri forest have now been developed (Beck 1989).

McCaw and Burrows (1989) have reviewed the historical patterns of fire use in jarrah forest and report a change from frequent low intensity fires before European settlement, to intense fires following agricultural development and early uncontrolled exploitation of the forest for timber. A policy of fire exclusion was introduced with the Forests Act in 1919 and continued until 1954 when controlled burning for broadscale fuel reduction was developed. This latter policy has been modified to account for selected animal species (Christensen 1983), dieback disease (Shea *et al.* 1981; Burrows 1985), silvicultural requirements (Stoneman *et al.* 1989) and the recognition of fire exclusion areas (Christensen and Abbott 1989).

Several authors have discussed the issue of fire periodicity in the karri forest (Christensen 1972; Talbot 1973; White 1977; Underwood 1978; and Christensen and Annels 1985). Although a detailed examination has yet to be undertaken, fire in the karri forest appears to have been less frequent than in the jarrah forest and some areas may have escaped fires for 30 years or more (Christensen and Abbott 1989).

About 70 per cent of south-west forests are prescription burnt on rotation for fuel reduction. Sections of the northern and central parts of the Darling District have been burnt regularly for about 30 years and the southern areas for about 20 years. The length of time between burns where fuel is to be reduced is based on accumulation rates and averages five to six years for jarrah and seven to nine years for karri forests (Christensen and Abbott 1989). Other areas are burnt less frequently and Christensen and Abbott (1989) listed 42 areas from which fire is excluded as a deliberate policy. National parks and nature reserves include areas of no planned burn, intermediate frequency and fuel reduction (see for example, Smith *et al.* 1990). More than 50 per cent of the public land in the Darling District has not been burnt for more than six years (Muller[1] personal communication 1990).

The high fuel accumulation in karri regrowth (Sneeuwjagt and Peet 1979) prompted the development of techniques for conducting the first fuel reduction burns in young karri regrowth. McCaw (1986) found that such burns could be initiated after 14 to 20 years, depending on site quality, with minimal damage to regrowth.

VEGETATION AND FLORISTICS

Short-lived species that regenerate from soil-stored seed have been studied in detail, owing to their dominance in the understorey of the karri and adjacent forest. Christensen and Kimber (1975) found that certain species were prolific seeders with large seed reserves in the soil. They also found that heat stimulated germination of seed of some legumes and

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that very little germination of those species occurred without heat treatment. However, seed was killed where litter was deep, and seed at shallow depth. In addition, moisture in the fuel or soil has an insulating effect which reduces heat penetration and thus germination. In the field, germination was improved by burning under dry soil conditions but soil temperatures varied greatly within a few centimetres. This variability on a micro habitat scale was used to explain the persistent germination of fire-sensitive species (e.g. *Bossiaea laidlawiana* and *Crowea angustifolia* var. *dentata*) following repeated fires at frequent enough intervals to prevent seeding of the regenerated plants.

Several shorter-lived species of legumes reached maturity and commenced seed production within three years and then quickly senesced (Skinner 1984). The longer-lived legume species *Acacia pentadenia* and *Bossiaea linophylla* had similar seed production patterns. *Bossiaea laidlawiana* and the dominant Rhamnaceae species, *Trymalium floribundum*, did not reach full seed production by age eight. Few differences in seed production were noted with site.

Baird (1988) studied *Banksia seminuda* ssp. *seminuda*, a fire sensitive obligate seed regenerator with a long juvenile period. He found that the juvenile period of *B. seminuda* varies between five and thirteen years and recommended that the frequency of fires in the watercourse habitat of the species be reduced either by excluding fires from those sites or by ensuring the burn rotation interval was more than five years.

Peet and Van Didden (1973) studied the impact of prescribed burning on 'understorey shrubs' near Manjimup. They revealed marked changes in structure as the species recovered from the effects of fire. Persistent species, regenerating from lignotuberous root-stock, recovered rapidly, while 'fireweeds' regenerating from seed regained their former level in cover contribution only in the third year following fire. Sneeuwjagt (1971) described understorey fuels in karri forest using the Levy point technique (Levy and Madden 1933) and determined structural types based on the dominant species comprising the understorey.

Christensen and Kimber (1975) concluded that in karri forest, the mosaic effect of prescribed burning ensures the perpetuation of existing plant communities and that species numbers increased after fire in true and marginal wet sclerophyll forests.

The popularity of *Boronia megastigma* in the wildflower industry prompted a preliminary investigation into the ecology of this species. Christensen and Skinner (1978) suggested that a prescribed burning regime of periodic spring burns in the forest would be of greater benefit to this species in the long term than would more intense autumn burns on the same rotation. This is because, under a spring burning regime, the moist gully habitat of Boronia generally remains unburnt for one or more rotations. Christensen and Skinner (1978) concluded that Boronia sites should remain unburnt for 10 to 15 years and that picking was detrimental to the survival of Boronia plants.

INVERTEBRATES

Most studies on the effects of forest management practices on invertebrate communities in the Warren Subdistrict have concentrated on the effects of fire. These include work by Springett (1976, 1978, 1979), Koch and Majer (1980), Majer and Koch (1982) and Curry *et al.* (1985). These studies have all been in karri forest but have been relatively brief and lack detailed taxonomic treatment.

Majer and Abbott (1989) reviewed the influence of disturbance on the soil and litter invertebrates of the jarrah forest and considered that they were well understood, though conclusions reached have not always been in agreement. The impact of burning on these animals has important implications for the rate of nutrient cycling because many help regulate the decomposition of litter. There is a trend, eastwards from the areas of highest rainfall, of slower litter fauna recovery following disturbance (Majer 1985; Majer and Abbott 1989) which needs to be accounted for in studies of the impact of prescribed burning on decomposer systems.

In jarrah forest, Springett (1976, 1979) found that soil fauna levels had not recovered in the five to seven year interval between prescribed burning treatments, while Abbott (1984) found all but three taxa recovered in density within three years following fire. The importance of taxonomic resolution is critical in studies of the effects of disturbances on invertebrate communities. Conflicting conclusions may also be a consequence of the sampling methods used. Campbell and Tanton (1981) were critical of Springett's (1976) work on the grounds of poor sampling and extraction techniques.

The high rainfall zone of the Warren Subdistrict provides habitat for many species of relict invertebrate taxa. For example, Kendrick (1978) and Solem *et al.* (1982) described new species of extant and extinct snails from the Leuwin Naturaliste Ridge and suggested that those groups of species with poor colonization ability may be vulnerable to disturbance. It has been suggested that certain Gondwanan spider relicts evolved in a rainforest environment and are therefore endangered by the frequent occurrence of fire in their environment unless the time of burning coincides with their dormancy (Main 1987, 1991). Although fire may be a rare event in such sites, occasional high intensity fires are likely in areas of high fuel accumulations such as the karri/tingle forests where many of these Gondwanan relicts occur. Relict spider species such as *Chasmocephalon* sp., *Dardarnus* sp., *Baalebulb* sp. and *Moggridgea tingle* (Fig. 8) have persisted in the relatively non-seasonal climate and moist protected habitat of the red tingle (*Eucalyptus jacksoni*) forest, restricted to a small area (< 6000 ha) near Walpole (Smith *et al.* 1990).

HERPETOFAUNA

The level of understanding of the herpetofauna in Western Australia has increased significantly in the past 20 years. There has been an increase in collections and surveys, a doubling of the species described or recognized (How *et al.* 1987) and a large increase in the number of publications (Daze 1984). Most early collectors avoided the subject and there is a poorly documented and recorded sub-fossil fauna. Hence it is difficult to evaluate historical changes in the herpetofauna. Apart from the *Geocrinia rosea* complex, there is little documented evidence on the effects of disturbance on reptiles and amphibians in the Warren Subdistrict.

The *Geocrinia rosea* complex includes four species of frogs exhibiting direct development of the eggs that are restricted to permanently damp sites in the lower south-west of Western Australia (Roberts *et al.* 1990; Wardell-Johnson and Roberts 1991). Two of these have been described only recently (Wardell-Johnson and Roberts 1989; Roberts *et al.* 1990). Wardell-Johnson and Roberts (1991) suggest that the rarity of their habitat requires conservative management practices in riparian habitat on private and public land alike and that protected riparian strips of native vegetation will be necessary to ensure their survival in the agricultural landscape (Fig. 9).

Figure 8
Moggridgea tingle , a relictual species of spider occurring only in the tingle forest near Walpole (Barbara York Main).

BIRDS

Christensen and Kimber (1975) studied the effects of a high intensity fire in summer, on birds (and mammals) occupying a 40 ha stand of karri regrowth which had not been burnt for 20 years and an adjacent control block. They reported that their results were difficult to interpret owing to large seasonal fluctuations of itinerants. Nevertheless, sightings declined in all strata immediately after the fire but recovered to above pre-fire levels in the lower three levels (four levels were studied) within five months. Christensen *et al.* (1985) concluded that the effect of fire on individual species was in general inversely proportional to their main foraging height in the foliage profile and is largely dependent on the intensity of the fire.

The two main layers forming the karri forest (understorey and overstorey, Fig. 10) support different components of the avifauna. High populations of a few species of residents occur in the homogenous and insulating structure of the understorey (Wooller and Brooker 1980; Wooller and Milewski 1981; Wardell-Johnson 1984, 1985). Four species (Golden

Figure 9
Remnant habitat of the endangered frog *Geocrinia alba* near Witchcliff. Over 70 per cent of the populations of this locally endemic species occur on private land

Whistler, *Pachycephala pectoralis*; White-browed Scrub-wren, *Sericornis frontalis*; White-breasted Robin, *Eopsaltria georgiana*; and Red-winged Fairy-wren, *Malurus elegans*) are common long-lived residents of the understorey with high annual survival rates of adult birds (Brown *et al.* 1990). Variability in the flowering of the canopy influenced the numbers of individuals (but not the number of species) present and seasonal differences in bird populations were greatest in the canopy (Wardell-Johnson 1984, 1985).

Wooller and Brooker (1980) report recovery of bird populations following a low intensity prescribed burn carried out in spring in Treen Brook Forest Block. They concluded that the composition of the bird community in the understorey was largely unaffected by prescribed burning and that many individuals remained in the area after such a fire.

Tingay and Tingay (1984) concluded that prescribed burning in spring in mature forest initially causes a short-term decrease in total abundance of birds followed by an invasion of small insectivorous species which prefer low, dense vegetation. Populations of these species were found to decline as the shrub stratum regenerates. Other species were found to increase in abundance until species number and abundance were largest in forest burnt six years previously. Thereafter, species number and abundance declined. However, some species were more common in long-unburnt forest.

Studies on the effects of disturbance on single species of birds in the Warren Botanical Subdistrict include Brown and Brown (1980), who examined co-operative breeding in robins of the genus *Eopsaltria*, and Rowley *et al.* (1988) who examined the ecology and breeding biology of the Red-winged Fairy-wren. Rowley *et al.* (1988) suggest that prescribed burning during spring may affect the Red-winged Fairy-wren because it is more abundant in areas not burnt for a long time and because of its poor colonizing ability.

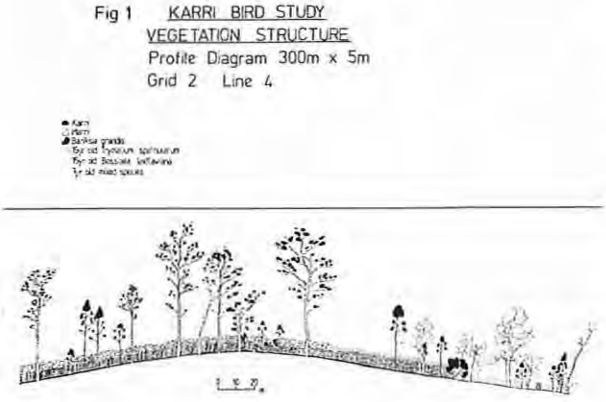

Figure 10
Profile diagram of mature karri forest in Grey Forest Block near Pemberton (Grant Wardell-Johnson).

MAMMALS

Research on the effects of fire on vertebrates in the Warren Botanical Subdistrict was initiated by Christensen and Kimber (1975). Their study reports on the effects of a prescribed fire, of wildfire intensity, on small mammals (and birds) in karri forest in Warren Forest Block near Pemberton. Three species of small mammals were regularly caught in 35 months of trapping, the House mouse (*Mus musculus*), Southern Bushrat (*Rattus fuscipes fuscipes*), and Mardo. House mice colonized the burnt plots within five months, reaching high levels in the year after the fire. No captures were reported 28 months after the fire. Recolonization by the bushrat started in the moist sites after 11.5 months and in more open understorey after 22.5 months. Mardos caught occasionally before the fire were not caught again during the 31 months following the fire. Christensen and Kimber (1975) present other trapping evidence to suggest that the Mardo may be more common in areas which have not been burnt for a decade or more and have high accumulations of litter and dead plant material. However, Mardos are also common in open woodland in Dryandra Forest and rare in karri/tingle forest regardless of biomass accumulation (Wardell-Johnson and Nichols 1991).

Wooller *et al.* (1981) and Wooller *et al.* (1984) have studied the Honey Possum (*Tarsipes rostratus*), in relation to season and nectar availability near Albany. The Honey Possum is considered vulnerable to habitat change brought about by dieback disease caused by the fungus *Phytopthora cinnamomi* or by fire regimes that disfavour key nectar producing taxa (for example, species of the families Proteaceae, Myrtaceae, Epacridaceae).

Christensen and Kimber (1975) report that the Quokka returns rapidly to swamp habitat after it has been burnt, but used it only as a feeding ground for at least a year. A large resident population was established within five years of fire. Christensen and Kimber (1975) suggest that there is some evidence for a population decline at 12-year-old sites and desertion of sites 15 years after fire. However, Quokkas are found in several places in the karri forest which have not been burnt for at least 20 years (Christensen unpublished data). Christensen (1980a) provided a detailed study of the ecology of the Woylie and Tammar in relation to fire in the neighbouring

Menzies Subdistrict. Neither species occur in the Warren Subdistrict.

The number of sightings of Western Grey Kangaroo (*Macropus fuliginosus*) and Brush Wallaby in relation to period since previous fire are presented by Christensen and Kimber (1975). Increased numbers of both species were evident in more recently burnt habitats. The Brush Wallaby was at the time considered common and almost half as many sightings of this species are reported as for the Western Grey Kangaroo.

Christensen and Kimber (1975) concluded that no one prescribed burning regime will encourage maximum population levels of all the mammal species in an ecosystem. They argued that prescribed burning should result in a mosaic effect if it is to simulate the uncontrolled fires and aboriginal burning of the period before colonization by Europeans. Infrequent high intensity fires and very frequent fires were considered factors unlikely to favour a burning mosaic.

Naturalized species

The present landscape in Australia is thought to have resulted from at least 40 000 years of human interaction (Singh *et al.* 1981). European settlement of Australia over the past 200 years has resulted in widespread and rapid modification of the environment, even in areas of retained native vegetation subject to minimal management. These changes are at least in part a result of the introduction, colonization and naturalization (Groves 1985) of a much greater range of species than before this period. The resultant mixture of native and introduced plant species that may exist in a metastable state has been termed synthetic vegetation (Bridgewater 1990). The term may be usefully applied to the biota in general and we may consider synthetic communities. Hence we include all introduced species that have become successfully established.

Though the Warren Subdistrict includes a more intact vertebrate fauna than possibly any other district or region in Western Australia, excepting the north-west Kimberley (Burbidge and McKenzie 1989), significant changes have occurred in species and community composition. Several animal introductions are now dominant components of the terrestrial fauna of the Subdistrict, for example, the Red fox, Domestic cat, European rabbit (*Oryctolagus cuniculus*), Black rat (*Rattus rattus*), House mouse, Laughing Kookaburra (*Daclo novaeguineae*) and European honey bee (*Apis mellifera*).

Other introductions are not yet well established but may become dominant components in the Subdistrict, for example, feral pig (*Sus scrofa*). The predicted expansion of the range of the Marine toad (*Bufo marinus*) (Sabath *et al.* 1981) and several species of birds (e.g. Common Starling, *Sturnus vulgaris*), and insects (e.g. European wasp) includes the south-west of Western Australia. None of these species is considered to be an ecological specialist and all except the Marine toad originate in climatic regimes similar to that in south-western Australia. Some species, such as the House mouse, are most common in communities subject to disturbance although this species is well established throughout south-western Australia.

The demise of many ground dwelling vertebrates in native vegetation has been largely attributed to the introduced Red fox which has also had considerable influence on the recovery of native fauna following disturbance (Christensen 1980b; Kinnear 1989; Kinnear *et al.* 1989; Friend 1990). The control or eradication of this species is considered the highest priority in fauna management in south-western Australia, along with the prevention of further introductions and expansions in the ranges of other introduced species, such as the Common Starling.

About 10 per cent of Australia's total of 15 000 to 20 000 species of vascular plants are introduced (Michael 1981). Although less taxa of weeds occur in the Warren Subdistrict than the Perth region (547 of 2057 (26 per cent) for Perth compared with 319 of 1947 (17 per cent) for Warren, Hopper *et al.* this volume), some of them pose major problems to conservation managers (Anon. 1987; Smith *et al.* 1990). For example, granite outcrop communities of the Porongurup Range, rich in endemic species, have been successfully invaded by *Plantago lanceolata, Cirsium vulgare, Briza* spp. and *Trifolium* spp. following disturbances such as grazing (Hopper *et al.* this volume). Similarly, *Rubus discolor, Rubus selmeri* and *Solanum nigrum* are dominant species along some creeklines in the karri forest (Fig. 11).

Major sources of weeds include rubbish tip sites (e.g. *Zantedeschia aethiopica, Ricinus communis* and *Solanum nigrum*), settlements (e.g. *Leptospermum laevigatum, Boralea pinnata* and *Eucalyptus sieberi*) adjacent agricultural land (e.g. *Pennisetum clandestinum, Ehrharta calycina* and *Trifolium* spp.) and revegetation schemes (e.g. *Acacia dealbata, Pinus radiata*). Griffin (1985) related the presence of weeds to the proximity of farmland and frequent fires in Smiths Brook Nature Reserve. However, the role of fire in either enhancing or minimizing invasions in native plant communities will depend on the physiological properties of both the native community and the invading organisms, and the fire regime (Christensen and Burrows 1986). Generally, frequent fire favours resprouting perennials over non-resprouting species, disadvantages species which

Figure 11
Blackberry (*Rubus* sp.) infestation along the Warren River near Pemberton. Blackberry is capable of becoming a dominant structural component of the flora in some riparian zones in the karri forest (Grant Wardell-Johnson).

rely solely on on-plant storage of seed, promotes herbaceous over woody plants, promotes grasses and forbs over dicotyledons, creates pure stands and reduces subsequent fire intensity (Vogl 1977). Frequent fire or other disturbance also favours aggressive introduced species and allows wind-dispersed species to establish. In this context, the word 'frequent' means less than a five-year interval. Many introduced species are able to resprout from rootstocks, and thereby survive a wide range of fire regimes (Christensen and Burrows 1986). Many introduced species, particularly grasses, are also aided in their spread and establishment by fertilizer drift from agricultural land.

Plant disease caused by introduced (and native) pathogens are perhaps the most serious disturbances to remnant native vegetation in the south-west. Dieback disease caused by the soil-borne pathogen *Phytophthora cinnamomi* has been present in the jarrah forest since the 1920s (see Shearer and Tippett 1989 for a review). Most of the information relating environment to disease development has come from research in the high rainfall zone of the Dale Botanical Subdistrict. However, the impact of plant disease in heath communities may be more serious than previously envisaged and major structural changes can occur (Wills 1992; Fig. 12).

Regeneration, dieback and site productivity were discussed by Strelein (1988) in the management of 17 site types in southern jarrah forest. All but one site type (type Q - high quality forest on fertile well drained loams) have some susceptibility to dieback disease, suggesting that logging operations could be more problematic than in the northern jarrah forest owing to the presence of high water tables and a longer period of moist soil conditions. However, climatic differences between northern and southern areas result in shorter periods of suitable soil temperatures and soil moisture levels which are necessary for activity of the fungus (Christensen 1975).

Although much is known about *Phytophthora cinnamomi* in jarrah forest, relatively little is known about most of the other *Phytophthora* species, or about *P. cinnamomi* in any other communities. *Phytophthora citricola* probably has the greatest potential to compound the problem of dieback disease caused by *P. cinnamomi* in the jarrah forest (Shearer and Tippett 1989). Other species may be serious pathogens in other environments. Air dispersed canker-causing fungi, such as *Botryosphaeria ribis* (Old *et al.* 1990), have severe impact in sandplain communities (Fig. 13) well outside the karri forest.

Figure 12
Impact of dieback disease caused by *Phytophthora cinnamomi* in a sandplain community near Walpole (Grant Wardell-Johnson).

CURRENT RESEARCH

Research relevant to disturbance ecology in the Warren Subdistrict is currently being undertaken within CALM's Biogeography, Native Forest Silviculture, Economic Entomology, Fire, Fauna and Flora research programs.

Figure 13
Impact of air-borne canker disease on a sand plain community at Black Point (Ray Wills).

Current disturbance research continues to concentrate on karri and jarrah. The effects of thinning and fertilizing on stem diameter, height, crown and branch development in karri regrowth forest in community types Stoate, Shea and Annels (Inions *et al.* 1990) is currently being examined by Hewett[2] (personal communication 1990). She is recording the cover-abundance of all vascular plant species in thinning experiments. Experiments to examine aspects of establishment, scrub control, site factors and fertilizing, spacing and establishment in disturbed jarrah sites are also currently in progress (Strelein[3], personal communication 1990).

Growth rate, crown recovery, bark thickness changes, and stem damage in response to fire in karri regrowth is currently being studied by McCaw[4] (personal communication 1990). Hewett (personal communication 1990) is examining similar responses in community type Stoate.

The incidence of borer (*Tryphocaria acanthocera*) infestation in karri forest is also being examined. The external symptoms of infestation of karri by borer and the incidence of infestation of borer in even-aged karri are currently being examined by Abbott[5] (personal communication 1990). The impact of wood-rotting fungi on karri regrowth in a variety of community types is currently being investigated by Davison[6] (personal communication 1990).

Current studies on economic entomology in the jarrah forest of the Warren Subdistrict include research on the impacts of jarrah leafminer and gum leaf skeletonizer on the growth of jarrah (Abbott, personal communication 1990). Abbott is also comparing effects between the northern and southern jarrah forest and examining the effects of fire on these insects. Farr[7] (personal communication) is examining fecundity and spatial distribution of gum leaf skeletonizer in relation to jarrah in the Warren Subdistrict.

Wardell-Johnson is examining floristics within an age series in a single community type to determine whether plant species composition and cover in the karri forest is related to age since disturbance. A complete list of vascular plant species has been gathered from 95 permanently located 400-m^2 quadrats, within a single community type (Shea) (Inions *et al.* 1990).

Current research on birds in the Warren Botanical Subdistrict includes an eight-year study in permanent sites in karri forest in Gray Forest Block. Following a two-year period of calibration (see Wardell-Johnson 1985), sites were either burnt, small gap (3 ha) felled, clearfelled or left as controls. Mistnetting, censusing and foraging observations were used to study the birds for five years after the operations. The structure and floristics of the vegetation are also assessed annually. Bird community composition is similar in small (3 ha) and large areas of retained mature forest and these areas have an important influence on the species composition of adjacent regrowth (Wardell-Johnson, unpublished data). Understorey bird species rapidly colonize regrowth as its structure develops.

The same methods of censusing birds employed in Gray Forest Block were used in a survey of the Walpole-Nornalup National Park. Survey sites were chosen according to different burn ages in all major vegetation structural types in the area. Since starting the work in December 1985, a wildfire burnt 34 of the 67 survey points in the Hilltop section of the Park, allowing the response of the bird community and terrestrial vertebrate fauna following wildfire to be assessed in comparison with sites that were not burnt. This work also addresses edge effects and structural changes to the vegetation. Measures of the vegetation

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[4] W.L. McCaw, Department of Conservation and Land Management, Manjimup.

[5] I. Abbott, Department of Conservation and Land Management, Como.

[6] E. Davison, Department of Conservation and Land Management, Como.

[7] J. Farr, Department of Conservation and Land Management, Manjimup.

structure and floristics were made each year for two years.

Serventy[8] (personal communication 1983) censused a 10.4-ha study site in a 105-ha coup of jarrah-marri forest near Pemberton between 1976 and 1982. The stand was heavily selection cut in 1978. Total numbers of individuals and species declined following logging but regained pre-cutting levels four years later (1982). Species using the outer foliage, mid-storey and understorey remained lower by 1982, while birds inhabiting the shrub layer increased. The lack of an untreated control limited the applicability of the results obtained, although they may be used as comparative data for other studies.

Research is in progress to determine whether hollows are limiting in the recolonization of regenerating karri coupes by Mardos and to investigate the effects of karri forest management techniques on hollow-nesting animals. Nest boxes are used in both studies though in the latter study only six species have so far used the boxes. Nest box studies are of limited use in tall forest as many species nest above the height where nest boxes can feasibly be located. Similarly, very large nest sites are required by some species.

A detailed study of the distribution and commercial utilization of Boronia species will enable the development of a wildlife management program for species subject to commercial exploitation (Hopper,[9] personal communication 1989).

Monitoring of long-term responses of plants to fire regimes commenced with work by Christensen in 1971 who aimed to remove *Bossiaea laidlawiana* and *Crowea angustifolia* var. *dentata* from an area of mixed jarrah and karri forest near Manjimup using fire of varying frequency. A total of six burns were completed in the treatments burnt at three-year intervals by 1986, but both *C. angustifolia* and *B. laidlawiana* were still found to be dominant (Christensen and Abbott 1989). The frequently burnt (three-year intervals) treatments were also found to have the highest number of species of plants.

Recent studies by Burrows include spring and summer burning of three, six and 12-year rotations in four study locations. These studies aim to provide biomass accumulation rates, complete species lists and structural data over a range of associations and fire regimes. Four main study locations were chosen: an area of karri forest at Strickland Road; low scrub and sedgeland sites of the Pingerup landform unit (Churchward *et al.* 1988); open jarrah forest at Perup; and an area of jarrah forest in the Blackwood Plateau. The latter two sites are in the Menzies Botanical Subdistrict.

The response to fire of all vascular plant species recorded in 145 permanent plots is being examined in the Walpole-Nornalup National Park. A total of 770 species have so far been collected. Monthly revisits to the quadrats has enabled the provision of a more complete species list for the quadrats, the recording of flowering status of all vascular plant species encountered and an understanding of the response of each species to disturbance. This study will provide similar information on fire in community type Wallace (Inions *et al.* 1990) in the Walpole area as research on community type Shea (Inions *et al.* 1990) in the Manjimup area.

Research to examine the ecological effects of an intense fire in karri forest was established by Christensen in 1971. Trapping continued until 1983, providing 12 years of trapping data on the effects of fire on populations of small mammals. Population fluctuations of the Southern Bushrat were found to override the long-term effects of a single fire on this species.

Research on plant disease in the southern forest and coastal areas of Western Australia is now in progress (Wills[10] personal communication 1990). Long term monitoring plots are being established and dieback vulnerability is being assessed on a species by species basis·

Current research on introduced predators emphasizes the severe impact of the fox on native fauna. Algar[11] (personal communication 1989) is researching its ecology and biology to find an efficient method of control. His work includes sites in the Darling District (e.g. Perup Nature Reserve where it is associated with the long-term monitoring of the Woylie population: Burrows[12], personal communication 1990) but not the Warren. Studies designed to enable the eradication or long-term effective control of the fox will be necessary for the future management of ground dwelling vertebrates generally in Australia. Christensen conducted a preliminary study of the Quokka in karri forest.

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Predation by foxes and the scarcity of study animals led to the termination of this work. The possible influence of disturbance (e.g. roadworks and felling operations) on predation of vulnerable species like the Quokka requires investigation.

Collections of the Warren Subdistrict flora are ongoing and continue to contribute to the knowledge of the naturalized flora of the area.

HIGH PRIORITY ADDITIONAL RESEARCH REQUIREMENTS

Most ecological research on the effects of disturbance on wildlife of the Warren Botanical Subdistrict has been in karri forest in the Nornalup system. Exceptions include studies in the Walpole-Nornalup and Mt. Frankland National Parks by Wardell-Johnson and along the south-coast between Albany and Busselton by Gibson[13] (personal communication 1990). We believe an expansion of such research into surrounding community types is needed, both in terms of single species studies and site-based ecological surveys. It is essential to understand the karri forest in the context of the Warren Botanical Subdistrict as a whole. Hence an understanding of the overall pattern of the biota for the Subdistrict is required.

Site-based ecological research should be expanded independently to the siting of specific coupes to provide a baseline understanding of the pattern of karri forest biota. A 'pre-logging survey' approach to ecological research is not considered to be a profitable use of limited resources. Rather, an investigation should be made which emphasizes particular groups or taxa.

The poor taxonomic base of invertebrate work urges that high priority be given to such work in the Warren Botanical Subdistrict in general and the karri forest in particular. Experimental work to examine the effects of disturbance on invertebrates will establish those vulnerable taxa and sites in need of special consideration in management operations and allow operations to be designed accordingly. Curry and Humphreys (1988) provide some insight into this problem for taxa caught in malaise and light traps but they did not study recovery after regeneration. Considerable additional work is required, including studies on ground dwelling invertebrates. Further experimental work should be accompanied by considerable taxonomic work. Groups that include relictual taxa of poor colonization ability are in particular need of study. Sites in high rainfall zones in karri-tingle forest may be expected to harbour such species.

Research on the growth of karri should now be directed to regeneration in areas where it grows in association with marri and jarrah (e.g. community-type Stewart, Inions *et al.* 1990). Some of the sites currently logged for jarrah are difficult to regenerate (Strelein 1988; Bradshaw[14], personal communication 1989). The most urgent silvicultural research on jarrah in the Warren Subdistrict is concerned with regeneration and establishment techniques on these sites. The identification and mapping of these sites may require further community study.

The opening of the canopy in thinning operations may extend the burning season in karri regrowth stands (McCaw, personal communication 1990). However, accumulations of logging debris may pose a considerable fire hazard. Fire behaviour is well understood in young unthinned stands and should now be directed to thinned stands.

While it is well established that stand structure is critical to the organization of species and communities at any one place or time (Recher 1969, 1971), temporal and spatial scales must be considered in examining the effects of broadscale management operations on wildlife (Recher *et al.* 1987). Rotation length and conservation strategies with respect to retention of mature forest are important in this context (see Fig. 14). For example, two forms of conservation strategy exist in the karri forest; discrete areas of forest reserved from logging such as national parks and nature reserves; and unlogged areas in any forest block (blocks average about 5000 ha). A series of corridors of forest are left unlogged on a block by block basis forming a network of corridors of road, river and stream zones which connect with National Parks and Nature Reserves. Continuing research into the optimal distribution of these and their long term management (Wardell-Johnson *et al.* 1991) is desirable.

Research is also required on the influence of individual mature trees (e.g. habitat trees) on particular species or communites in the context of clearfelling and other logging methods. Detailed studies to determine an age series of vertebrate and invertebrate (particularly those species of poor colonization ability) fauna in areas of intensive harvesting remain a high priority. The study of bird

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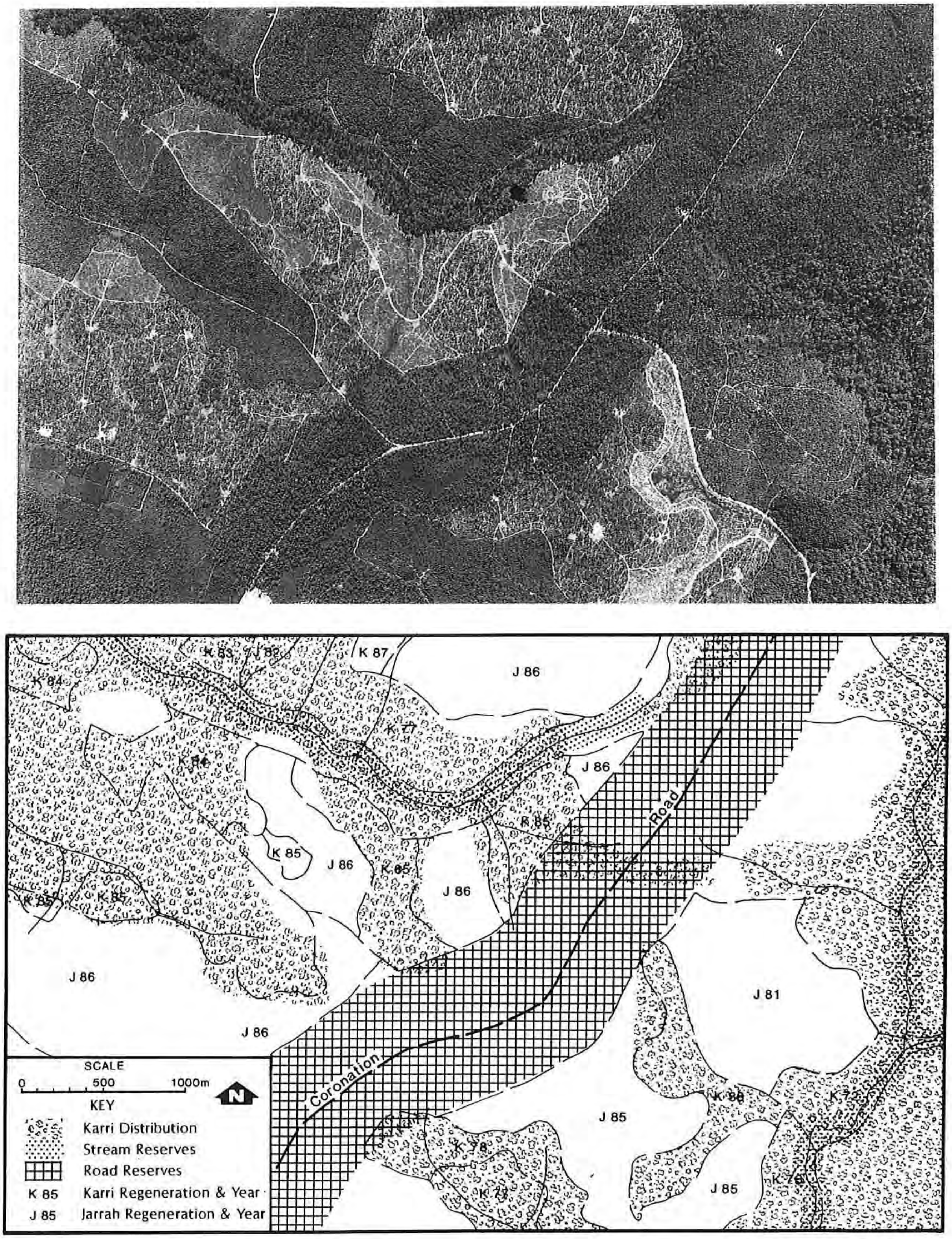

Figure 14
The spatial pattern of harvesting and regeneration in an area near Pemberton. Note the pattern of the distribution of road, river and stream zones.

communities in areas of different management history in the karri forest is a logical extension of the studies in permanent plots described earlier, as the composition of the community occurring in the overstorey is affected by structural change for a considerable time (Tingay and Tingay 1984).

The program approach adopted by the Department of Conservation and Land Management should be used in further research in the Warren Subdistrict to allow the establishment of integrated studies on the most pressing areas remaining. For example, data on the distribution of most forest vertebrates are inadequate to define changes in status or be useful for predictive modelling of the effects of disturbance. Models of the impact of disturbance will follow from integrated studies such as those on fire in the Eyre District (Friend,[15] personal communication 1989).

Considerable research is now available or in progress on plant species known to regenerate effectively from soil-stored seed or that resprout from lignotubers following disturbance. However, not all species regenerating by other means (e.g. canopy stored seed) have been studied. For example, little is known of the response to disturbance or management needs of most fire-sensitive obligate seed regenerators with a long juvenile period (e.g. *Banksia seminuda* ssp. *remanans, Banksia verticillata* (Fig. 15), *Banksia occidentalis, Grevillea brevicuspis, Hakea lasianthoides, Hakea oleifolia, Dryandra sessilis, Persoonia microcarpa, Petrophile diversifolia* and *Isopogon formosus*). Similarly, no work is currently in progress on the response of bulbous, cormous or herbaceous species to disturbance, particularly spring burning, that occurs when many of these species are actively growing.

Research emphasis on fauna is required on species considered to be vulnerable to disturbance. Species of animals of poor dispersal ability (e.g. snails, some Arachnidae), of low population densities (e.g. Brush Wallaby, Western Ringtail), requiring patchily distributed habitat (e.g. Quokka) or requiring habitat components that require a considerable time to form (e.g. hollow nesting species such as Baudins Cockatoo, *Calyptorhynchus baudinii*, McKenzies Bat, *Falsistrellis mckenziei* and the Barking Owl, *Ninox connivens*) must be studied in relation to the effects of forest use and management.

The timing and cause of formation of hollows in karri, marri and jarrah trees requires further research. Research by Mackowski (1984) and Faunt[16] (personal communication 1990) would serve as a useful basis for the detailed study of these eucalypts. In view of the long term required for hollow formation in karri (Wardell-Johnson, unpublished data), marri (Wardell-Johnson, unpublished data), and jarrah (Inions 1985) trees, the guild of birds and mammals requiring hollows as nesting sites should be examined in detail and preferably as single species studies.

Dieback disease has the potential to lead to major long term structural and floristic changes in a wide range of vegetation communities in the south-west leading to changes in animal communities. The monitoring of fauna in association with the establishment of quadrats for floristic study will allow predictions of the future of dieback-infected communities.

Changes in biomass and cover that may result following disturbance may also lead to changes in microbial communities (Shearer, this volume). The importance of microflora and fauna in the functioning of ecosystems has not been studied in the Warren Subdistrict and will require urgent attention.

There has been considerable research on the effects of disturbance on wildlife of the karri forest. Priority areas for further research relevant to disturbance ecology in the Warren Subdistrict include:

- development of an improved understanding of the overall pattern of the biota;
- the impact of disturbance on invertebrate communities and the identification of indicator taxa;
- the development of predictive models of the impacts of disturbance on vertebrate communities;
- the growth, development and decline of key plant species (e.g. karri) including the process of hollow formation;
- the control of key naturalized species (e.g. *Phytophthora*, Red fox) and the maintenance of community processes;
- the role of microflora/fauna communities; and
- vulnerable species (including Gondwanan relicts) and their habitats.

A multidisciplinary approach to process oriented research is recommended in disturbance ecology.

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Figure 15
A stand of *Banksia verticulata* four years after fire in the Walpole-Nornalup National Park. This species has on-plant seed storage and is sensitive to fire (Grant Wardell-Johnson).

Regional staff are now involved in the establishment and monitoring of ecological reference sites applicable to operations involving disturbance.

ACKNOWLEDGEMENTS

We thank Ian Abbott, Janet Farr, Penni Hewett, Angas Hopkins, Lachlan McCaw, Roger Underwood and Ray Wills, for comments on an earlier draft of this manuscript. Joanne Healey typed a draft of this manuscript, Michelle Pree drafted Figure 1, Paul Davies and Annette Turpin drafted Figure 8, Barbara York Main provided Figure 9 and Ray Wills provided Figure 14.

We also thank Malcolm Howes and Marianne Lewis for editorial comments.

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CHAPTER 3

Impact of Timber Harvest and Regeneration on Nutrition of the Jarrah and Karri Forests of South-West Western Australia

John Bartle and John McGrath

Abstract

Full rotation length nutrient balances are estimated for karri (*Eucalyptus diversicolor* F. Muell.) and jarrah (*E. marginata* Donn ex Sm.) forests of the south-west of Western Australia. These balances take into account the storage, input and export of the six major nutrients (nitrogen, phosphorus, potassium, sulphur, calcium and magnesium) over rotation lengths of 100 years in karri and 120 years in jarrah. Although only approximate, these balances provide a useful framework for evaluating management impacts on the biological cycling of nutrients. Examination of the balances reveals that nutrient export at harvest is small in relation to nutrient storages and inputs over the whole rotation. Further research needs are identified and discussed.

INTRODUCTION

The concept of nutrient balance provides a systematic framework within which to evaluate impacts of management on forest nutrition. The nutrient balance is the net result of gains by and losses from the forest ecosystem over a given period. When related to the gross storage of nutrient in the ecosystem a simple indication of the likely importance of the nutrient balance can be gained. This can be used to guide research or modify management.

The usefulness of nutrient balance analysis is usually constrained by the availability of data. However, sufficient data are available for the karri (*Eucalyptus diversicolor* F. Muell.) and jarrah (*E. marginata* Donn ex Sm) to provide reasonable full rotation length estimates of nutrient balance for the major nutrients.

Therefore, this paper reviews the currently available data on karri and jarrah forest nutrition and assembles it into a nutrient balance context. Within this systematic framework the role of timber harvest and regeneration, fire and fertilizer application are discussed and research needs identified.

REVIEW OF CURRENT KNOWLEDGE

Balances are assembled for the six major nutrients (nitrogen, phosporus, potassium, sulphur, calcium and magnesium) for typical karri and jarrah forest sites (Table 1). The karri data come mainly from Hingston *et al.* (1979), and are for a pure karri stand on a red-earth categorized as White community type by Inions *et al.* (1990). The only jarrah data available are for the northern jarrah forest (Hingston *et al.* 1980/81, 1989): these data are the average of 13 yellow sandy lateritic gravel sites and are assumed to be representative of southern forest jarrah on similar soils. Both types are assumed to receive 1100 mm rainfall per year, and to have rotation lengths of 100 and 120 years respectively. Each balance includes a full harvest of sawlogs and chips, followed by regeneration.

Soil Nutrient Storage

The soil nutrient storage data in Table 1 are separated into available and total categories. In presenting this data Hingston *et al.* (1979), indicate that the criteria used to segregate the available and total categories have been developed for agricultural applications. For this purpose the available category includes only nutrients in readily extractable forms and at shallow depth (less than 1 m), and which have been found to be correlated with response to fertilizer in agricultural plants.

Table 1
PROJECTED NUTRIENT BALANCES (kg ha^{-1}) FOR KARRI AND JARRAH FOREST

	Nitrogen		*Phosphorus*		*Potassium*		*Sulphur*		*Calcium*		*Magnesium*		*Comments*
Component	*karri*	*jarrah*	*karri*	*jarrah*	*karri*	*jarrah*	*karri*	*jarrah*	*karri*	*jarrah*	*karri*	*jarrah*	
Storage													
soil total	NA	NA	1718[a]	725[b]	6214[a]	5861[b]	2906[a]	1296[b]	10224[a]	5508[b]	5346[a]	2200[b]	Soil storages are for top 90 cm in karri and top 100 cm in jarrah for the size fraction $^{-2}$mm
soil available	7439[a]	1536[b]	54[a]	14[b]	471[a]	110[b]	260[a]	118[b]	3827[a]	11[b]	1027[a]	240[b]	
biomass	249[a]	327[b]	20[a]	14[b]	264[a]	311[b]	24[a]	60[b]	737[a]	353[b]	151[a]	182[b]	Excluding roots
litter	224[a]	48[b]	7[a]	2[b]	32[a]	8[b]	28[a]	8[b]	396[a]	90[b]	60[a]	20[b]	Litter age 6 years (since previous fire)
TOTAL	7912	1911	1799	755	6981	6290	3218	1482	15184	7137	6584	2642	
Inputs													
atmospheric	50[c]	60[c]	30[b]	36[b]	500[d]	600[d]	700[d]	840[d]	900[d]	1080[d]	700[d]	840[d]	For karri rotation of 100 years and jarrah rotation of 120 years
weathering	NA	NA	8[e]	8[e]	900[e]	1080[e]	NA	NA	600[e]	720[e]	200[e]	240[e]	Weathering 0.01 mm of new baseme rock per annum
fixation	1000[f,g]	1080[g,h,i]	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	
TOTAL	1050	1140	38	44	1400	1680	700	840	1500	1800	900	1080	
Outputs													
fire	1282	480	NA	NA	NA	NA	159	80	NA	NA	NA	NA	Burning rotation 6 years in jarrah 9 years in karri
drainage	nil	nil	nil	nil	185[j]	216[j]	422[j]	493[j]	528[j]	616[j]	1188[j]	1386[j]	
harvest	108[a]	101[b]	10[a]	4[b]	147[a]	90[b]	8[a]	19[b]	550[a]	62[b]	102[a]	34[b]	204 t ha^{-1} for karri 120 t ha^{-1} for jarrah. Assumes total bark is taken in harvest
TOTAL	1390	581	10	4	332	306	589	592	1078	678	1290	1420	
Balance: Inputs minus outputs	-340	559	28	40	1068	1374	111	248	422	1122	-390	-340	

Source

a. Hingston *et al.* (1979) single red earth site table III page 145
b. Hingston *et al.* means of 13 sites in northern jarrah
c. Bell and Barry (1980) and O'Connell (1985)
d. Hingston and Gailitis (1976)
e. Hingston (1977)
f. Grove and Malajczuk (1980)
g. O'Connell and Grove (1986)
h. Hingston *et al.* (1982)
i. Grove *et al.* (1980)
j. Loh *et al.* (1983)

It is likely that trees can utilize forms of nutrients not in the available category, albeit slowly, but in amounts which would be important over a full rotation (Crane 1978; Turner 1981). The lateritic jarrah forest soils have 50 per cent of gravel greater than 2 mm diameter in the top 1 m of the profile. Hingston *et al*, (1980/81) indicate that this fraction can contain as much or more nutrients than the fine earth (less than 2 mm) fraction, but, on the assumption that it is physically and chemically unavailable to roots, it has been excluded from consideration as a nutrient source for trees. However, physically isolated nutrients may be accessible to the long-term activity of micro-organisms symbiotically associated with tree roots.

The soils occupied by forests of the south-west of Western Australia, especially the jarrah forest, have unusually deep weathered profiles. Dell *et al.* (1983), suggest that the whole weathered profile, with depths averaging 20 m (Johnston *et al.* 1980), is exploited by roots. The phenomenon of salt storage arises from the near equivalence of rainfall and evapotranspiration. Under this regime the trace of oceanic salt in the rain accumulates as storage deep within the soil profile rather than being discharged into the drainage systems. Total soluble salt storage in the 1100 mm rainfall zone averages 7.5 x 104 kg ha^{-1} (Stokes *et al.* 1980). Some 10-15 per cent of this salt storage (10 000 kg ha^{-1}) comprises the soluble nutrient ions potassium, sulphate, magnesium and calcium (Loh *et al.* 1983). It is also likely that there are additional less soluble forms of these and other nutrients in the deep subsoil which may be available to tree roots.

Hingston *et al.* (1979) can therefore be strongly supported when they state that their data should not be considered an absolute measure of the proportion of nutrient capital which is available to forest plants. It should also be considered an underestimate of the size of both total and available storages.

Biomass and Litter Storage

The biomass and litter nutrient data are from Hingston *et al.* (1979) and Hingston *et al.* (1989).

The karri stand studied consisted of 36-year-old even-aged regeneration with 6-year-old understorey. It is the same site for which soil nutrient storage data were presented. The biomass included surface dead wood but excluded roots. However, some of the root nutrients would have been included in the soil nutrients. The biomass total at this site was 236 t ha^{-1}, of which 184 t was bolewood, 20 t bark, 20 t leaves and branches, 10 t understorey and 2 t was dead wood.

For jarrah the data presented are the means for 13 sites on lateritic soils. No detailed breakdown of biomass and litter data was presented by Hingston *et al.* (1989). However, details were provided for one of the sites in Hingston *et al.* (1980/81) which had nutrient contents similar to the means of the bulked data and has therefore been taken as representative biomass breakdown data. Total biomass was 269 t ha^{-1}, of which 186 t was bolewood, 37 t bolebark, 41 t leaves and branches, and 5 t understorey. Dead fallen wood amounted to 130 t ha^{-1} but since this is a variable component of the nutrient store it has been excluded.

Nutrient Inputs

Estimates of the annual rates of nutrient input from the atmosphere, weathering and nitrogen fixation are available. These have been summed over a nominal rotation length (i.e. 100 years for karri, 120 for jarrah) to provide rotation length input estimates.

Atmospheric input can arrive as dry fallout or can be contained in rainfall. For the southern forests of Western Australia the major source of atmospheric input is oceanic. Input varies with rainfall and the following levels, appropriate to 1100 mm rainfall, have been used: 9 kg calcium, 7 kg magnesium, 7 kg sulphur and 5 kg potassium (Hingston and Gailitis 1976) 3.6 kg nitrogen (Bell and Barry 1980) and 0.3 kg phosphorus (Hingston *et al.* 1989).

As previously discussed it is likely that the entire depth of the weathered profile in the jarrah and karri forests is exploited by roots. Nutrients released from current weathering of the igneous basement rocks may therefore be available for uptake by roots. Hingston (1977) based his speculative estimates of nutrients released by weathering on an annual attrition of 0.01 mm thickness of rock over unit area i.e. 270 kg ha^{-1} annually.

Fixation of nitrogen in biomass is a major input, although it is difficult to quantify. Fixation occurs by symbiotic association of micro-organisms with plants, especially in the family Leguminosae, in the species *Macrozamia riedlei* and the genus *Allocasuarina.* Other non-symbiotic forms of nitrogen fixation also occur. The rate of fixation depends on the density, vigour and age of the fixing species. This in turn is linked to fire (especially to the role of fire in the regeneration of legume species), moisture availability and fertility. All of those factors may display great variation in time and space.

In parts of the karri forest the leguminous species *Bossiaea laidlawiana* is a dominating understorey component: it is capable of sustaining high levels of nitrogen fixation even in mature stands. Grove and Malajczuk (1980) observed fixation rates of 6-8 kg ha^{-1} annually at age 5-8 in regenerating forest and rates of 14 kg ha^{-1} annually in mature forest with 11-year-old understorey. This strong nitrogen-fixing capability is reflected in the competitive advantage of *B. laidlawiana* over the non-leguminous codominant understorey species *Trymalium spathulatum.* Biomass and nitrogen content data for growth of these species over time is presented in Table 2 from data in Grove and Malajczuk (1985).

Table 2
BIOMASS AND NITROGEN CONTENT OF KARRI UNDERSTOREY

Age (years)	*Biomass* *t* ha^{-1}		*Nitrogen* *kg* ha^{-1}	
4	2.1	1.3	14	4
8	17.5	12.7	103	30
11	23.7	12.3	129	24

a. B.l is *Bossiaea laidlawiana*
b. T.s is *Trymalium spathulatum*
(from: Grove and Malajczuk 1985)

O'Connell and Grove (1986) found that the environment within moist karri litter favours non-symbiotic fixation of nitrogen, and estimated rates of 2 kg ha^{-1} annually.

To project a rotation length nutrient balance an annual nitrogen input of 10 kg ha^{-1} has been assumed (Table 1). It is stressed that this integration over space and time must be considered very approximate.

In jarrah forest a greater diversity of legume species occurs and other non-legume nitrogen fixing species are also more numerous. The jarrah forest has also attracted more research work. Some discrepancy between nitrogen fixation rates for the same species has been reported by different workers. Hingston *et al.* (1982) reported rates of 7 kg ha^{-1} for a dense young stand of *Acacia pulchella*, whereas Hansen *et al.* (1987) reported the same species averaging less than 1 kg ha^{-1} annually over six years. The explanation for this discrepancy may be in part the difference in density, age and size of the stand but it may also reflect the capacity of jarrah forest legume species to opportunistically utilize soil nitrogen, or conversely, where conditions are favourable, to fix very large quantities of nitrogen. For example, in rehabilitated bauxite pits in the jarrah forest local legume species dominate the early stages of revegetation which, with generous application of phosphorus fertilizer, can produce up to 100 t of biomass and increase the nitrogen in the system by 207 kg ha^{-1} annually over 5.5 years (Koch 1987). Nitrogen fixation rates averaging 6 kg ha^{-1} annually have been reported for a stand of *Macrozamia reidlei* averaging 0.14 plants per metre (Grove *et al.* 1980).

No data on the nitrogen input from *Allocasuarina* are available. Non-symbiotic fixation in litter is less than 1 kg ha^{-1} annually (O'Connell and Grove 1986).

For the projected nutrient balance in jarrah forest an annual average input of 9 kg nitrogen has been assumed (Table 1). As for karri such a projection over space and time must be regarded as very approximate.

Nutrient Export

Export of nutrients can occur through fire, drainage and harvesting.

Losses of both nitrogen and sulphur occur during fire. Some loss of phosphorus and other nutrients may also occur but this is not well documented. Litter and understorey biomass contain the nutrient storages most vulnerable to fire. Fire can range from intense wildfire to low intensity controlled burns used for reducing fire hazard. The consumption of fuels varies greatly within a burnt area. These factors make the estimation of average losses owing to fire extremely difficult. There may also be a net loss of nutrients in ash which is redistributed locally or regionally by turbulence during fire. Evans *et al.* (1976) reported that up to 4 per cent of ash may be transported over long distances in smoke plumes; the other 96 per cent is redistributed nearby.

Losses of nitrogen during fire in eucalypt forest have been measured by Raison *et al.* (1985). They found losses ranged from 50 - 75 per cent of the nitrogen in the burnt fuel. The losses of sulphur may be smaller in proportion (Raison *et al.* 1984) but a factor of 50 per cent is used for both nutrients for low intensity prescribed burns and 75 per cent for the karri regeneration burn.

The burning regime for karri forest includes the initial regeneration burn followed after age 20 by regular prescribed fires on a 9-year cycle to give some nine fires over the nominal 100 year rotation. The regeneration burn is assumed to consume all biomass and litter not harvested (although this would rarely happen). The nitrogen storage is 473 kg (biomass plus litter) less 108 kg (harvested bolewood), 75 per cent (i.e. 274 kg) of which is lost. Similarly for sulphur there is 52 kg (biomass + litter) less 8 kg (bolewood), 75 per cent (i.e. 33 kg) of which is lost. The loss owing to prescribed burning is 1008 kg nitrogen (50 per cent of 224 kg nitrogen in litter x 9 burns) and 126 kg sulphur (50 per cent of 28 kg sulphur in litter x 9 burns). The total losses are 1282 kg nitrogen and 159 kg sulphur (Table 1).

In jarrah the burning regime does not include an intensive regeneration burn, and prescribed burns are on a 6-year cycle giving 20 burns over a 120 year rotation. Total losses of nitrogen are 480 kg (50 per cent of 48 kg for 20 fires) . For sulphur, losses are 80 kg (50 per cent of 8 kg for 20 fires) .

These estimates are conservative in that no allowance is made for possible combustion of green biomass, especially the understorey where fine fractions (leaves and twigs) contain a disproportionate amount of nutrients and are readily burnt (Grove and Malajczuk 1985). On the other hand, prescribed burns do not always consume the entire litter layer. The fine layer near to mineral soil contains two thirds of the nitrogen and sulphur (O'Connell 1989) and is the fraction most likely to escape complete combustion. Also spring burns leave a mosaic of unburnt patches throughout the forest.

Drainage losses are considered to include output in streamflow only. Deep leaching of nutrients does not remove them from the root zone in karri and jarrah forest soils so that leached nutrients remain in the available storage category. Losses are estimated using streamflow salt concentration data from Loh *et al.* (1983), and streamflow volume data from the Steering Committee for Research on Land Use and Water Supply (1987). It is assumed that streamflow volume and salt concentration are increased for 10 years after harvest from 10 per cent to 20 per cent of rainfall and 200 to 300 mg L^{-1} respectively. It was assumed that no losses of nitrogen or phosphorus occur in streamflow.

Nutrient Balances

Table 1 indicates that harvesting losses are a small part of rotation length nutrient balance in both karri and jarrah production forests and are of no particular concern. The negative balances indicated for nitrogen and magnesium in karri, and magnesium in jarrah warrant further discussion.

In the case of the nitrogen balance in karri the available soil storage is probably enough to sustain productivity for several 100 year rotations. In the case of the negative magnesium balance in karri and jarrah, the available storage does not cover the deficit and a steep decline in productivity might be predicted.

Wise management of the nitrogen balance would demand that gains to the ecosystem should be arranged to comfortably cover losses owing to volatilization in fire. This matter has been analysed by O'Connell (1989). He suggests a lengthening of the prescribed burning cycle (from 9 to 13 years) and a lessening of fire intensity to protect the nutrient-rich lower layer of the litter from complete combustion. However, such a solution increases the risk of wild fire, which may not be acceptable. Another course explored by Grove (1988) is to enhance fixation by fertilization with phosphorus. He found that phosphorus application increased legume biomass and enhanced the competitive advantage of legumes over other understorey species, the reverse of what occurred with application of nitrogen fertilizer. These observations suggest an equilibrium between legumes/non-legumes and nitrogen/phosphorus in the ecosystem, which may be naturally self-correcting. As the soil nitrogen store is depleted a relative increase in the vigour of legumes occurs to correct the balance.

The negative magnesium balances indicated in Table 1 arise from inaccurate estimation of drainage losses rather than real nutrient imbalance. This occurred because fixed annual losses for two periods during the rotation were integrated to give a whole rotation loss. It is inappropriate to do this since large losses would not be sustained with the progressive depletion of the soil store. It is also clear that the total soil store is larger than that given for the upper 1 m of the soil profile in Table 1. This subsoil store could sustain considerable losses after disturbance to the water balance, analagous to those observed for salt, but losses would diminish steeply once the effect of the water balance disturbance had passed (Steering Committee for Research on Land Use and Water Supply 1987). The subsoil store may also be accessible to tree roots and so be involved in biological cycling. A more obvious reason why magnesium nutrition presents no concern is that total biomass requirement is well provided for by input from rainfall (Table 1).

Scope For Fertilizer Application

Attiwill (1979) has defined three phases of forest growth for eucalypts.

1 The establishment of photosynthetic and metabolic transport systems up to canopy closure. The rate of nutrient requirement during this stage is relatively high.

2 The heartwood formation stage, during which some nutrients are immobilized in heartwood, but large quantities of mobile nutrients (nitrogen, phosphorus, potassium) are remobilized during heartwood formation. Nutrient demands are reduced during this phase.

3 The maintenance phase during which forest mass is more or less stable, and nutrient cycling both internally and through the litter layer maintains the nutrient requirements of the trees.

Thus, it is during the first two of these stages that nutrient requirements are greatest. However, in a managed forest in which thinning occurs, it may be possible to increase growth rates during the third phase, as thinning effectively converts the forest back to the first phase where the photosynthetic and transport systems of the forest are not fully developed.

Abbott and Loneragan (1986) have reviewed the impact of fertilizer application on the growth of jarrah. Responses to nitrogen and phosphorus by jarrah seedlings grown in pots have been extensively documented. However, evidence of fertilizer responses by jarrah in the lignotuberous stage in the forest is lacking. Increased growth rates in pole stands of jarrah have been documented following the application of nitrogen, or nitrogen in combination with phosphorus and potassium, the responses being greatest in thinned stands (Kimber unpublished, quoted by Abbott and Loneragan 1986). The greater response in thinned stands is consistent with the hypothesis that if other non-nutritional factors (in this case water) are constraining growth, then responses to fertilizer will be limited. Currently no operational fertilization is carried out in the jarrah forest. However, considering the inherent infertility of the soils of the jarrah forest, there may be scope to improve the growth rate of jarrah, particularly in the higher rainfall areas, either during the early stages of growth or following thinning.

Seedling karri respond to nitrogen and phosphorus both in pots (Loneragan and Loneragan 1964) and in the field (Christensen 1974). Nitrogen and phosphorus are applied at planting to all karri seedlings planted in areas regenerated following clearfelling. Responses to nitrogen and phosphorus and a large positive interaction between nitrogen and phosphorus have been demonstrated for karri during the developmental stage of the forest (Attiwill's first growth phase) by Grove (1988). Although the response by karri to application of fertilizer following thinning has been studied in two separate experiments (Inions 1990), the design of these experiments precludes any definite conclusions being made about the possibility of the responses to nitrogen and phosporus being commercially significant. It is likely that wood production can be significantly increased by applying nitrogen and phosporus fertilizer to karri during the establishment and development phases, and following thinning.

CURRENT FERTILIZER RESEARCH IN JARRAH AND KARRI FORESTS

Departmental research into the nutrition of these forests consists of two projects examining the interaction between stand density (thinning) and fertilizer response. The experiments in the jarrah and karri forests have commenced and involve the monitoring of forest growth and tree and soil water relationships.

ADDITIONAL RESEARCH REQUIREMENTS

The major areas where insufficient information exists are:

1 Quantification of fertilizer response during the early growth of jarrah (from lignotuber to pole stages and karri seedlings to canopy closure).

2 Extension of the later age fertilizer experiments in jarrah and karri forests to a greater range of sites so that the range of responses to fertilization can be determined (i.e. determine the impact of climate, soil type and stand development on fertilizer response).

3 An understanding of the processes of nutrient cycling and the impact of fertilization on these processes in both the jarrah and karri forests. This would enable the development of efficient fertilization practices.

4 Determine the impact of fertilization on fuel loads in forests.

5 Determine the impact of fertilization on the occurrence of herbivorous and wood boring insects.

6 Determine whether the natural processes of regulation of the nitrogen economy in karri under prescribed burning are adequate to maintain the nitrogen status of the forest and whether it can be improved with phosphorus fertilization.

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CHAPTER 4

Review of the Impact of Logging and Regeneration on the Hydrology of the Southern Forest of Western Australia

Geoff Stoneman

Abstract

Research into both the short-term and long-term effects of logging in the southern forest of Western Australia on water quality and quantity began in 1973. Four major projects were undertaken.

Permanent clearing was shown historically to have led to increases in stream salinity in the intermediate and low rainfall zones (<1100 mm y^{-1}). Substantial increases in stream salinity following logging and regeneration were thought to be unlikely.

Vegetation cover returned to unlogged levels within 5 to 10 years for karri *(Eucalyptus diversicolor)* forest and 10 to 15 years of logging and regeneration for jarrah-marri (*E. marginata-E. calophylla*) forest. Groundwater levels rose for 2 to 4 years after logging and then started to decline. They should return to pre-logging levels by 13 years after logging and regeneration. Groundwater responses were much less in the low rainfall zone (<900 mm y^{-1}) than the intermediate and high rainfall zones (>900 mm y^{-1}). Stream salinities increased by 2 years after logging and regeneration and then declined. Stream salinity did not increase in the low rainfall zone. Stream sediment concentrations increased on catchments which had no stream buffer and were logged during winter. Sediment concentrations increased for 1 to 2 years following logging and then declined to pre-logging levels over the subsequent 3 years. Catchments which had a stream buffer and were logged during summer had no increase in sediment concentrations. Streamflow volumes increased for 2 to 3 years after logging and regeneration and then declined, and should return to pre-logging levels by about 12 years after regeneration. The increases in streamflow volumes were about 10 per cent of rainfall in the intermediate and high rainfall zones and less than 5 per cent of rainfall in the low rainfall zone.

Areas of continuing research and high priority additional research which is required are discussed.

PAST RESEARCH AND CURRENT KNOWLEDGE

In the late 1960s there was a move towards more intensive logging in the southern forest for various reasons (White and Underwood 1974; Bradshaw and Lush 1981; Borg *et al.* 1987b). Clearfelling (previously practised from the 1890s to 1940) was reintroduced in the karri *(Eucalyptus diversicolor)* forest and the successful regeneration treatments which occurred in the jarrah forests pre 1939 were re-instated. However, there was concern that clearfelling in the karri and intensive logging in the jarrah (*E. marginata*) forest might affect water quality (Environmental Protection Authority 1973; Forests Department 1973).

The Environmental Impact Statement on a proposed woodchipping industry in the Manjimup Woodchip Licence Area (Fig. 1) discussed risks to water quality (Forests Department 1973). It was decided that the industry should proceed but that research should be undertaken to quantify the effect on water quality (Environmental Protection Authority 1973).

In 1973 a Technical Steering Committee for Research on the Woodchip Industry was formed. The research program established by that committee is now the responsibility of the Steering Committee for Research on Land Use and Water Supply. This Committee is responsible to the State Government through the Coordinating Committee for Research on Land Use and Water Supply. Before establishing the research program no quantitative information was available on how logging affects stream salinity or stream sediment concentration. The research program was established to quantify both the short-term and long-term effects of the new logging operations.

The objectives of this paper are to (a) review research on the effects of logging on the hydrology of the southern forest, (b) describe research that is continuing and (c) describe high priority additional research that is required.

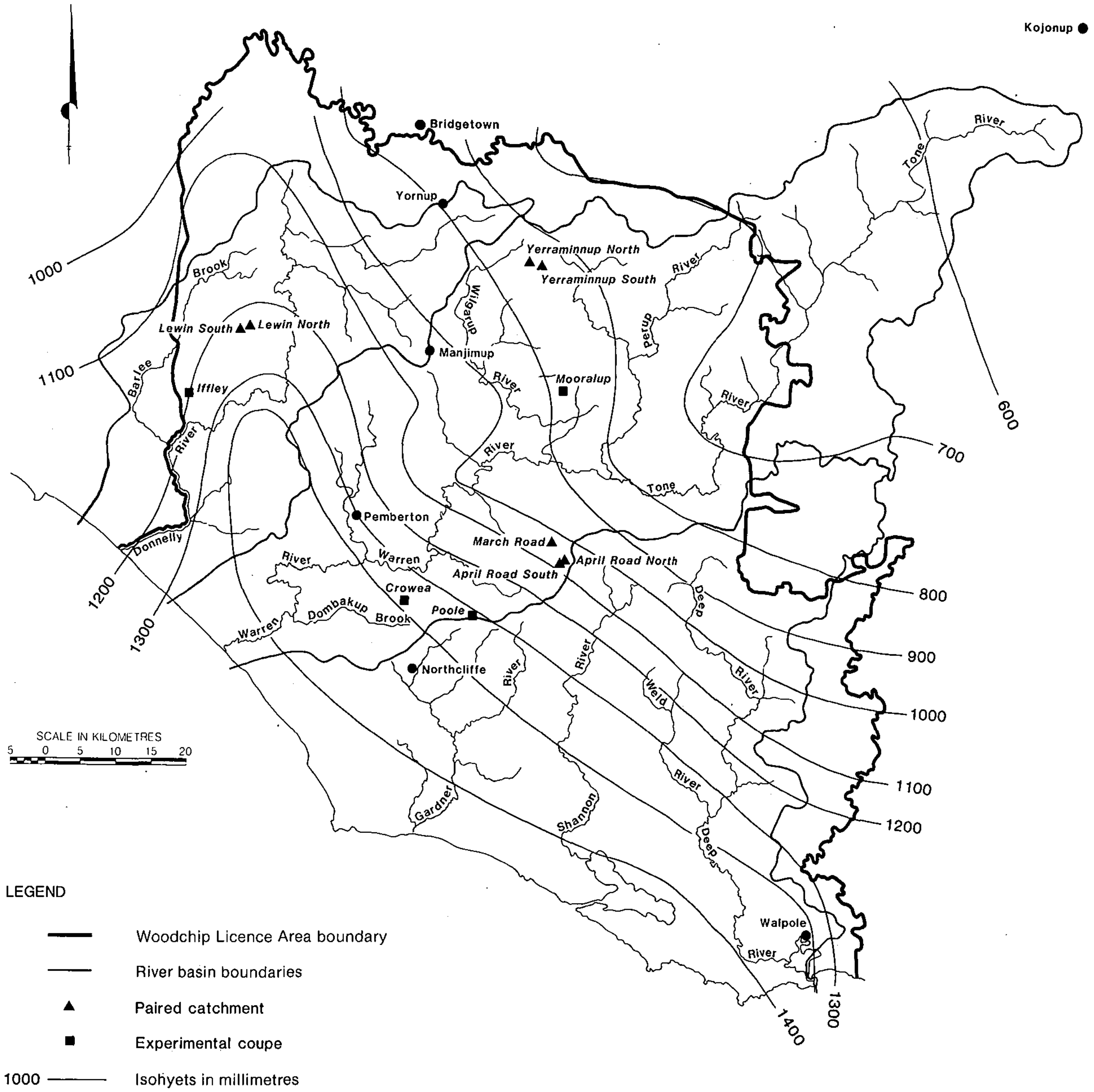

Figure 1

Location of paired catchments (Project 2) and experimental coupes (Project 4)

Research Program

The research program of the Steering Committee For Research on the Woodchip Industry involved four major projects:

PROJECT 1
Areas vulnerable to salinity increase were identified by (a) an extensive borehole drilling program of sampling for soil salt storage from 1974-1976 (Johnston *et al.* 1980), (b) an historical assessment of land clearing and its affect on salinity from records dating back to 1911 (Department of Agriculture 1974), and (c) stream salinity sampling in two major partly cleared catchments in the intermediate rainfall zone (900-1100 mm y^{-1}) during 1977 with the results correlated with land use and other catchment characteristics (Steering Committee 1980).

PROJECT 2
A detailed paired-catchment experiment was established to provide accurate information on changes caused by the proposed logging operations to both surface and groundwater hydrology (Steering Committee 1978, 1980, 1987; Borg *et al.* 1987a; Martin 1987). Seven experimental catchments in three groups covering the range of rainfall, landform, soils and forest types were established (Fig. 1). Measurements of rainfall, stream salinity and sediments, streamflow, groundwater level and groundwater salinity began in 1975 and data were analysed to 1985. Following a pre-treatment calibration, one catchment in each group was left as a control and the others were logged and regenerated in accordance with the current range of forest management practices. In this way, changes in streamflow volumes, salinities and sediment loads caused by the catchment treatment could be identified, independent of their large natural variations. In addition, about 10 to 12 groundwater bores on each catchment were established to measure the response of the groundwater system to the catchment treatment. Logging began in 1982 and most logging and regeneration operations were completed by the winter of 1983. Post-treatment results from three years of monitoring were analysed (Borg *et al.* 1987a; Martin 1987). Measurements of vegetation density were taken during 1986 in these catchments and on other sites with a range of times since regeneration to relate to the changes in catchment hydrology (Stoneman *et al.* 1988; Stoneman *et al.* 1989; Borg and Stoneman 1991).

PROJECT 3
Major rivers in the southern forest were monitored to measure any large-scale changes in water quality caused by logging. Daily sampling for sediment and salinity concentrations was carried out at 12 gauging stations, with varying proportions of forest and agricultural development, from 1975 up to 1986 for most stations. Data have been reported up to 1979. Historical records of stream salinity dating back to 1961 were also analysed (Steering Committee 1980).

PROJECT 4
Four operational coupes, selected as experimental catchments (Fig. 1), were monitored for one year prior to logging in 1977. Monitoring was not as thorough as in the Project 2 catchments and pre-treatment calibration was very limited. Measurements of rainfall, streamflow, stream salinity, groundwater level and groundwater salinity were taken from 1975 to 1985 (Steering Committee 1978, 1980, 1987; Borg *et al.* 1987b, 1988b). The nine years of post-treatment data that were analysed from these experimental coupes have provided valuable information, particularly on the longer term groundwater responses to logging (Borg *et al.* 1987b, 1988b). Measurements of vegetation density were also taken in these catchments as described above for Project 2 (Stoneman *et al.* 1988; Stoneman *et al.* 1989; Borg and Stoneman 1991).

Research Results

PROJECTS 1 and 3
Where salinity increases were reported within the Manjimup district, they were found to be associated with permanent clearing in the intermediate and low rainfall zones (<1100 mm y^{-1}) (Department of Agriculture 1974). This is distinct from timber harvesting operations which are followed by rapid regeneration of the deep-rooted vegetation. It was found that logging associated with the woodchip industry could have a deleterious effect on stream salinity only in areas where sufficiently large quantities of salt are stored in the landscape. Soil salt storage generally increases with decreasing mean annual rainfall (Fig. 2) (Johnston *et al.* 1980). Evidence from these and other studies in forest areas of the Darling Range (Dimmock *et al.* 1974; Stokes *et al.* 1980; Johnston 1981; Tsykin and Slessar 1985), have shown the most hazardous conditions for stream salinity occur in the lateritic soils of the low rainfall zone (<900 mm y^{-1}) where soil solute concentrations are >2000 mg L^{-1} total soluble salts (TSS). A moderate salinity hazard also exists in the intermediate rainfall zone (900-1100 mm y^{-1}).

Experience with Projects 1 and 3 indicates that over extended periods in the intermediate rainfall zone there has been some introduction of salt to rivers from areas of permanent clearing. However, results from these projects suggested that there would be no substantial increases in stream salinity following the logging and regeneration. The results also show that in the high rainfall zone (>1100 mm y^{-1}) woodchipping poses no salinity risks to the region's water resources.

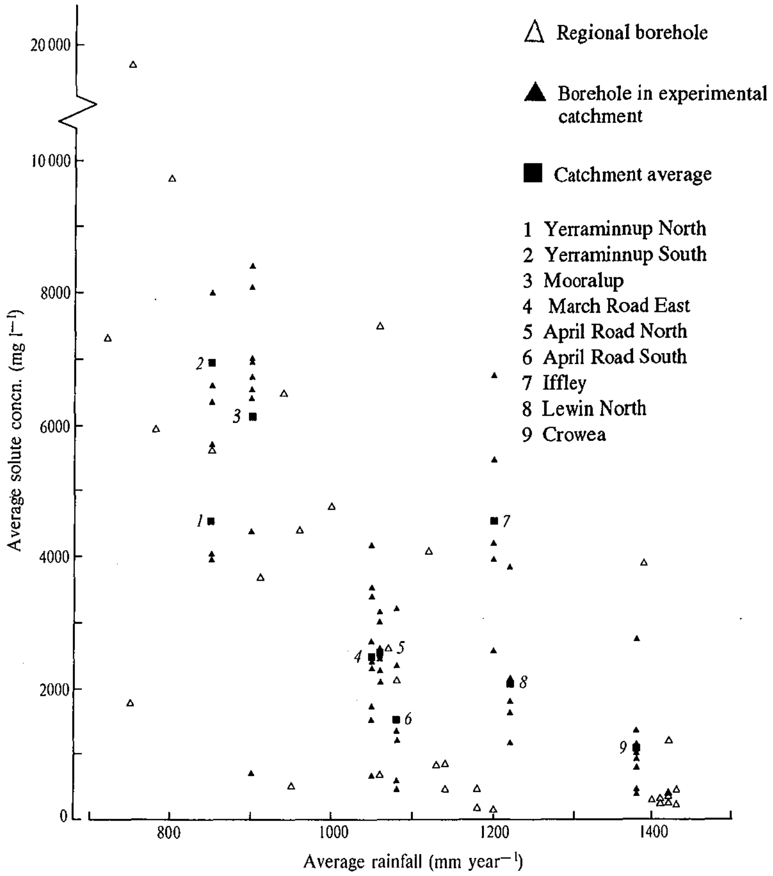

Figure 2
Soil solute concentration in relation to mean annual rainfall (from Johnson *et al.* 1980).

In forested areas in the low rainfall zone groundwater is generally characterized by high salinity (>2000 mg L^{-1} TSS) and great depth below the surface (usually >4 m and often >15 m) (Borg *et al.* 1987a, 1987b; Schofield *et al.* 1989). Except in very dissected landscapes, this saline groundwater needs to rise a great distance before it is able to enter the streams. This would normally take a long time with no forest cover (groundwater rises of 0.8 m y^{-1} were estimated by Loh and Stokes (1980) for a low rainfall zone area following clearing, and groundwater rises of 1.3 m y^{-1} were found following clearing of 54 per cent of a low rainfall zone catchment near Collie (Peck and Williamson 1987)), and much longer with a little forest cover (groundwater rises of 0.4 m y^{-1} followed strip and parkland clearing of 38 per cent of a low rainfall zone catchment near Collie (Peck and Williamson 1987)). On the other hand, in the higher rainfall zones the groundwater is fresher (<2000 mg L^{-1}), more widespread (>50 per cent of catchment areas) and closer to the surface, so it often flows direct to streams under natural forest conditions (Schofield *et al.* 1989).

PROJECTS 2 and 4

Rainfall ~ Rainfall for the study period (1975-1985) was 10 per cent below the long-term mean. The drier conditions are likely to have reduced the size of the hydrologic response to logging and regeneration, but not the general trends (Borg *et al.* 1987a, 1987b, 1988b).

Vegetation cover ~ The vegetation regenerated quickly in all rainfall zones (Fig. 3). In karri stands vegetation cover approached unlogged levels within 5 to 10 years and while actively growing to maturity would achieve higher densities than the original mature forest unless thinned. Jarrah-marri stands responded more slowly, reaching 90 per cent of unlogged cover in about 10 to 15 years (Stoneman *et al.* 1988).

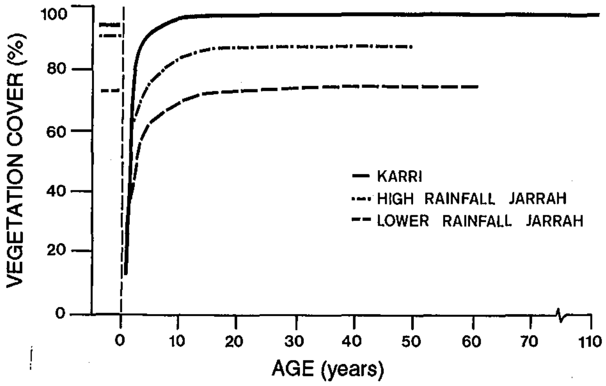

Figure 3
Relationship between vegetation cover and stand age for lower and high rainfall jarrah stands and karri stands (from Stoneman *et al.* 1989).

Groundwater Responses ~ In the Project 4 catchments, permanent groundwater levels (relative to control bores) rose for two to four years after logging and then started to decline (Fig. 4). The groundwater levels were still higher than pre-logging values eight years after logging, but were continuing to decline. Assuming (the then) current rates of decline, it was estimated to take a further five years for the permanent groundwaters to return to the level they would have been without logging i.e. 13 years after regeneration (Borg *et al.* 1987b, 1988b). Three years after regeneration commenced on the treated Project 2 catchments, groundwater levels approached their peak or began to decline on the jarrah-marri catchments but had yet to reach their peak on the karri catchments (Borg *et al.* 1987a).

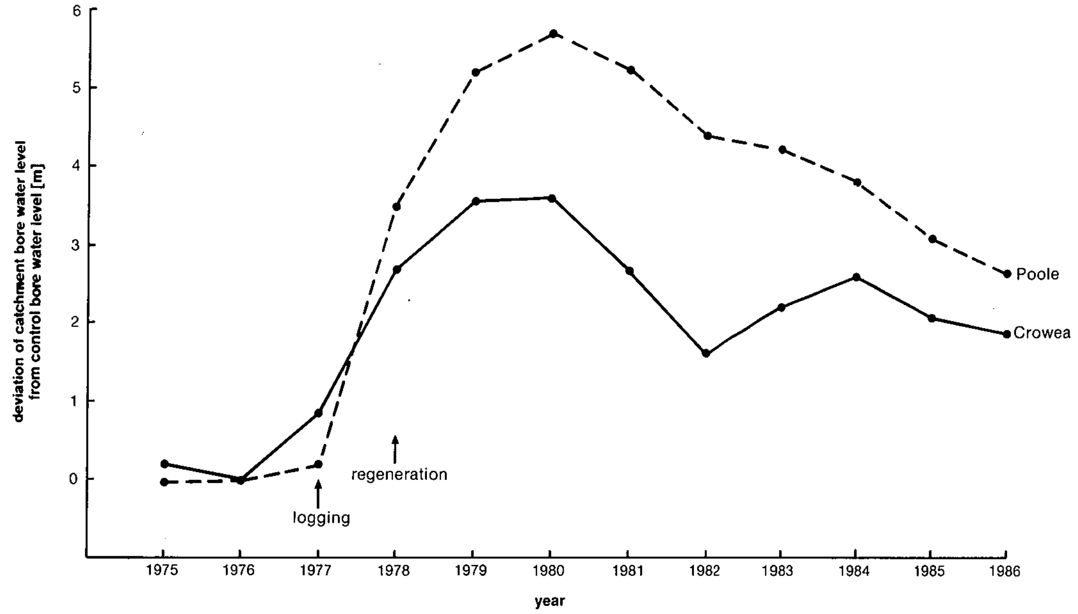

Figure 4
Changes in groundwater level relative to control boreholes following logging in two experimental coupes (from Borg *et al.* 1987b).

Results from both projects showed that groundwater responses to logging were much less in the low rainfall zone than in the intermediate and high rainfall zones (>900 mm y^{-1}) (Borg *et al.* 1987a, 1987b, 1988b). Changes in groundwater level were shown to relate to vegetation cover with the rise in groundwater level ceasing when vegetation cover reached the value for unlogged stands (Fig. 5) (Stoneman *et al.* 1989).

Stream Salinity Responses ~ Small increases in annual flow-weighted stream salinities of between 50 and 150 mg L^{-1} TSS occurred on most treated experimental areas (Fig. 6) (Borg *et al.* 1987a, 1987b, 1988b). However, all annual flow-weighted salinities remained below 500 mg L^{-1} TSS, the limit for high quality drinking water. In the Project 4 catchments, maximum annual stream salinities occurred two years after regeneration commenced, and salinities have since declined (Borg *et al.* 1987b, 1988b). In the Project 2 catchments, annual salinities were highest in 1985, two years after regeneration commenced (Borg *et al.* 1987a). The largest increase in annual flow-weighted salinities (about 150 mg L^{-1} TSS) occurred on a clearfelled catchment in the intermediate rainfall zone which did not have a stream vegetation buffer (Borg *et al.* 1987a). Salinity during periods of low flow on this catchment increased from 700 mg L^{-1} pre-logging to more than 1500 mg L^{-1} post-logging, although these values occurred mainly during the first few weeks of winter flow. Salinities in excess of 1500 mg L^{-1} occurred for over 10 per cent of the time of flow during 1985, but only represented 1 per cent of the flow volume (Borg *et al.* 1987a).

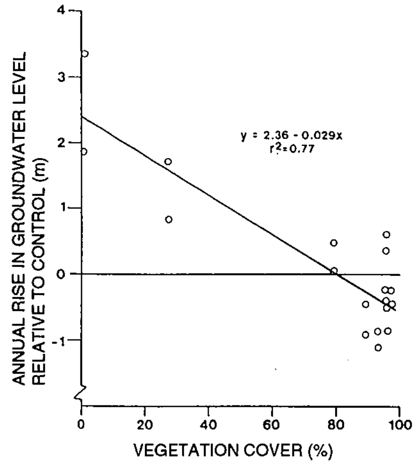

Figure 5
Relationship for karri forest between annual change in groundwater level relative to control boreholes and vegetation cover (from Stoneman *et al.* 1989).

Since the research program began there has been particular interest in the effect of the new logging strategies on water quality in the low rainfall zone where the soil salt storage is high (Environmental Protection Authority 1973). However, experimental results have shown there was no stream salinity increase in this area (Borg *et al.* 1987a, 1987b, 1988b). This is because recharge was small and the depth to groundwater was sufficiently large so that groundwater (the major source of salt) did not contribute to streamflow following logging (Borg *et al.* 1987a, 1987b, 1988b).

In the intermediate and high rainfall zones, groundwater contributed to streamflow before logging. Following logging in these zones, permanent groundwater levels rose and stream salinities increased, indicating an increase in the discharge of salts from groundwater to streams. Similarly, as groundwater levels began to fall following regeneration, stream salinities fell. It is expected that stream salinities will return to near pre-logging values (Borg *et al.* 1987a, 1987b, 1988b).

From a regional water resource perspective, the salinity increases observed are minor. However, the low flow salinities measured at greater than 1500 mg L^{-1}, if they persist for many weeks, could cause problems with small-scale public water supply systems based on low-volume storages. This problem can be overcome by appropriate design of vegetative stream buffers (Steering Committee 1987).

Sediment Concentrations ~ Stream sediment concentrations before logging were less than 5 mg L^{-1} in all monitored catchments (Steering Committee 1978, 1980, 1987; Borg *et al.* 1987a). Sediment concentrations increased on two of the four paired catchments and remained high for one to two years following logging, before declining to pre-logging levels over the next three years (Fig. 7) (Borg *et al.* 1987a). The catchments where measurable increases in sediment concentrations occurred had no buffer of streamline vegetation retained and were logged through the winter periods. The highest annual flow-weighted sediment concentrations were 38 mg L^{-1}. No sediment increases were detected on the catchments which had a buffer of streamline vegetation retained and were logged during dry summer conditions (Borg *et al.* 1987a). Less detailed monitoring of trials where stream buffer widths were reduced were carried out through 1985 and 1986 (Borg *et al.* 1987a). Reducing the width of existing river buffers from 200 m to 100 m, and reducing the width of existing stream buffers from 100 m to 50 m had no effect on water quality when logging took place over the summer (Borg *et al.* 1987a).

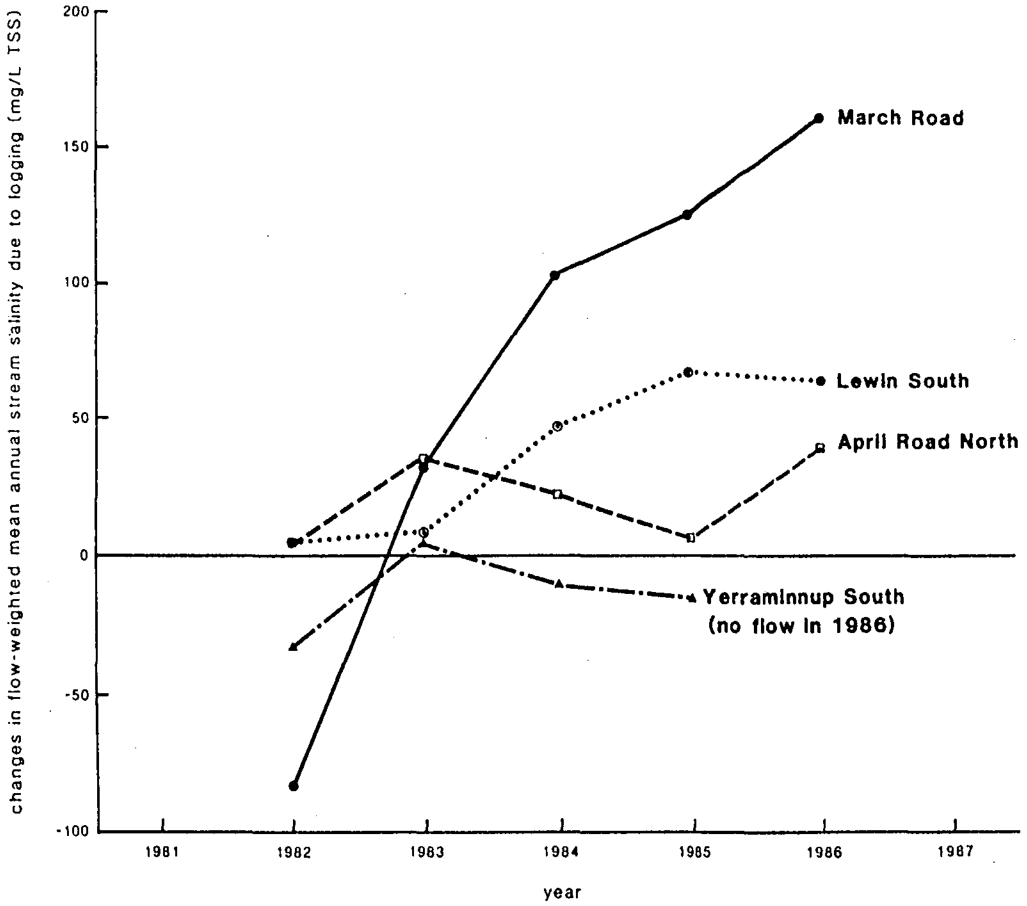

Figure 6

Changes in flow-weighted mean annual stream salinity in the four logged research catchments (from Borg *et al.* 1987a).

In a regional water resources context, the sediment increases were minor, caused in part by the practice of wide dispersal of the logging operations. These stream sediment increases would be of concern to a drinking water supply storage of small volume and short retention time as insufficient time would be available for turbidity levels to reduce in the storage (Steering Committee 1987).

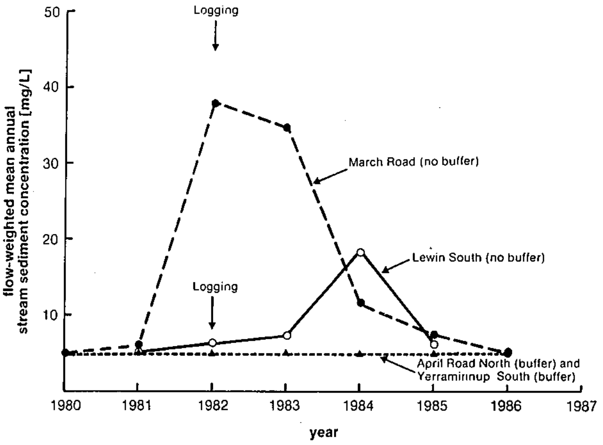

Figure 7
Changes in flow-weighted mean annual stream sediment concentration in the four logged research catchments (from Steering Committee 1987).

Water Yield ~ Streamflow volumes increased at all sites for two to three years after logging and then began to decline (Borg *et al.* 1987a, 1987b, 1988b). The Project 2 catchment results indicated a doubling of streamflow volumes in these early years (Borg *et al.* 1987a). In the high and intermediate rainfall zones, these increases were about 10 per cent of rainfall, whereas in the low rainfall zone the increase was less than 5 per cent of rainfall (Borg *et al.* 1987a, 1987b, 1988b). Results from the Project 4 catchments suggest that streamflows will return to pre-logging levels by about 12 years after logging and regeneration (Fig. 8) (Borg *et al.* 1987b, 1988b). If all stands were subsequently left unthinned (which is not proposed), a significant reduction in water yield could result (Borg and Stoneman 1991) over a period of the stand life.

CURRENT RESEARCH

The research effort has scaled down and emphasis placed on the most critical areas. Project 1 finished in 1981 and all but groundwater monitoring on the Project 4 catchments was discontinued at the end of 1985. The Department of Conservation and Land Management is responsible for monitoring these boreholes. The major stream sediment sampling program (Project 3) was discontinued on all but one stream at the end of 1986. The Water Authority of Western Australia is responsible for monitoring this stream. However, the program has not yet had time to fully quantify the hydrologic effects of timber harvest and regeneration. Only three years post-treatment data were available for the Project 2 catchments. This research, which is the reponsibility of the Water Authority of Western Australia, is continuing to:

1 quantify the longer term water yield and stream solute concentration changes and groundwater responses;

2 evaluate the effectiveness of phased logging and regeneration in minimizing stream sediment and salinity concentrations, and groundwater responses, by logging and regenerating of the stream reserve on April Road North catchment;

3 measure the long-term water yield response of regrowth karri forests at the Sutton Block catchments and determine appropriate thinning regimes.

HIGH PRIORITY ADDITIONAL RESEARCH

Additional work is needed to:

1 develop practicable means of assessing the local risks of salt mobilization and groundwater discharge in the intermediate and low rainfall areas. This is a responsibility of the Department of Conservation and Land Management;

2 develop appropriate stream buffer definitions and subsequent logging and regeneration prescriptions for stream zones based on the more accurate assessment of the local salinity and sediment risks. This should be a joint Department of Conservation and Land Management and Water Authority effort.

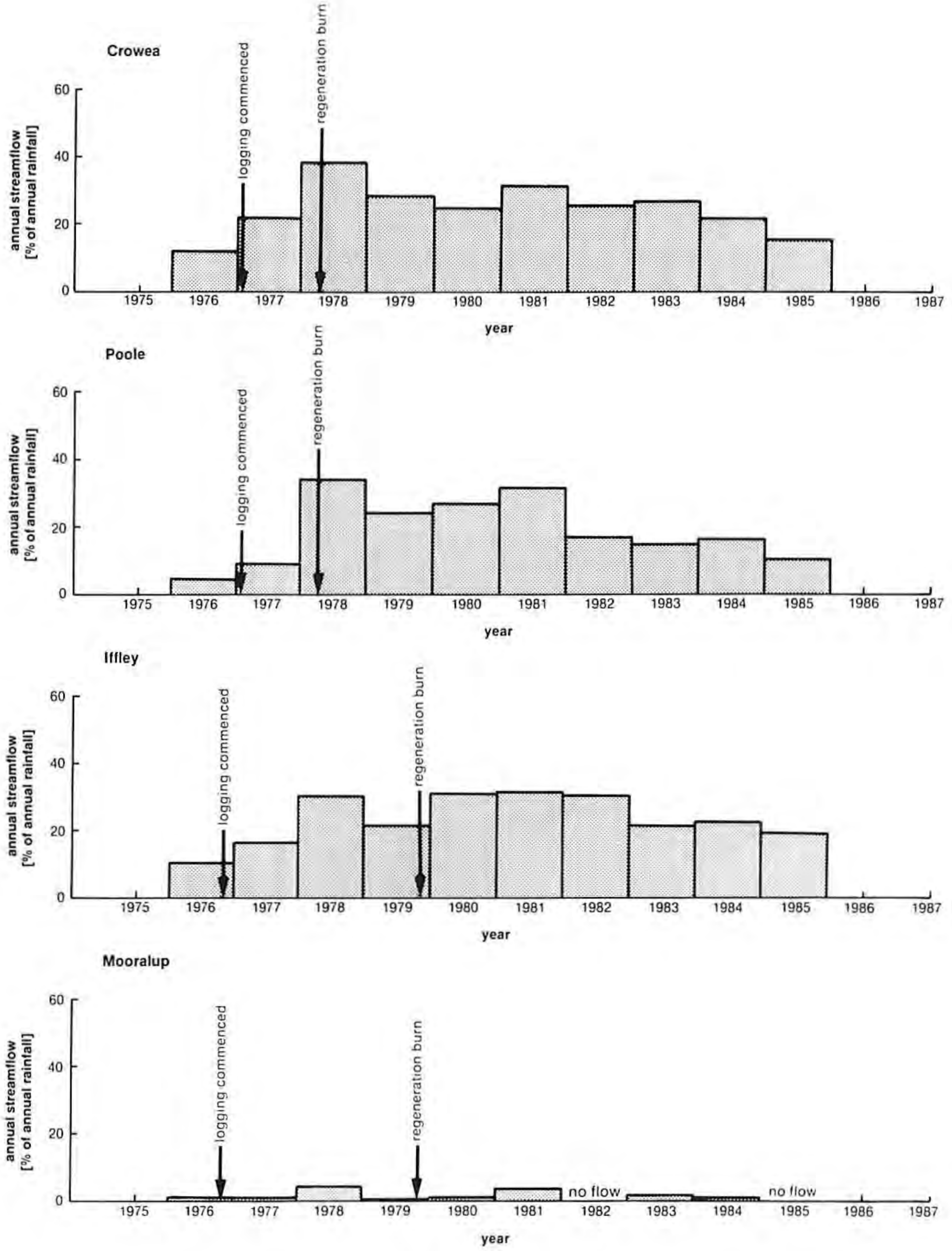

Figure 8
Annual streamflow in the four logged experimental coupes
(from Steering Committee 1987).

ACKNOWLEDGEMENTS

I thank Ian Loh for reviewing the manuscript and Marianne Lewis for editing it.

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CHAPTER 5

Ecological Implications of Insect Pests in Jarrah and Karri Forests

Ian Abbott

Abstract

Unlike most forested parts of the world, the hardwood forests of Western Australia did not experience insect outbreaks until 30 years ago, even though these forests have been commercially logged for up to 80 years.

This paper reviews what is known and what is not known about the three most important forest insect pests and their impact on conservation values and stand dynamics and productivity. These pests are: *Perthida glyphopa*, jarrah leafminer (Lepidoptera, Incurvariidae); *Uraba lugens*, gumleaf skeletonizer (Lepidoptera, Noctuidae); and *Tryphocaria acanthocera*, bullseye borer (Coleoptera, Cerambycidae).

Current research is summarized under five broad headings, and 12 additional high priority research requirements are briefly discussed.

INTRODUCTION

No serious insect problem was reported in jarrah (*Eucalyptus marginata*) forest until the early 1960s, when the feeding of jarrah leafminer (Table 1) caused extensive defoliation of jarrah crowns east of Manjimup. This infestation has gradually extended west and north, and at present occurs as far north as Collie.

A second problem arose in 1983 when outbreaks of gumleaf skeletonizer were first recorded in jarrah forest between Nannup and Walpole. Infestation by this species expanded within this region until 1985. It then declined and from 1989 has not been of concern.

The only insect problem known in karri (*E. diversicolor*) forest concerns the bullseye borer, the larvae of which bore within the bole of karri. Foresters became aware of this problem in 1967.

The relative importance of these species is shown by the most recent reliable estimates of areas infested: jarrah leafminer, 420 000 ha in 1976; gumleaf skeletonizer, 240 000 ha in 1986 (0 ha infested in 1991); bullseye borer, not known but probably not exceeding the area of even-aged regenerated karri stands (currently 40 000 ha). Leafminer infests 56 per cent of the high quality jarrah forest.

The maximal extent of recent infestations by leafminer and by skeletonizer is marked in Figure 1.

This paper updates Abbott (1985), and reviews and summarizes knowledge about these three insect species, discusses their impact in the forest, outlines current research conducted by CALM, and concludes with a catalogue of urgent research needs.

PAST RESEARCH AND CURRENT KNOWLEDGE

The three major insect pests in jarrah and karri forests have not been given the same degree of research attention. The oldest problem, jarrah leafminer, has been studied longest (CSIRO 1962 - 1988, Forests Department and CALM 1967 and 1984 to the present).

Jarrah leafminer

The CSIRO studies have yielded a detailed body of information on life history (Wallace 1970; Mazanec 1980, 1983, 1984a, b; Mazanec and Justin 1986), population biology (Mazanec 1978, 1981), natural enemies (Mazanec 1987, 1988a, 1990a, b), resistant trees (Mazanec 1974, 1985), chemical control (Wallace 1966), and the short term impact of infestation on wood increment (Mazanec 1974).

Table 1

TAXONOMIC DETAILS FOR INSECT SPECIES IN THE ORDER MENTIONED IN THIS PAPER

Venacular Name	*Latin Binomial*	*Order*	*Family*
Jarrah leafminer	*Perthida glyphopa*	Lepidoptera	Incurvariidae
Gumleaf skeletonizer	*Uraba lugens*	Lepidoptera	Noctuidae
Bullseye borer	*Tryphocaria acanthocera*	Coleoptera	Cerambycidae
-	*Diadegma* sp.	Hymenoptera	Ichneumonidae
-	*Chrysonotomyia* sp. C	Hymenoptera	Eulophidae
Leafblister sawfly	*Phylacteophaga froggatti*	Hymenoptera	Pergidae
Autumn gum moth	*Mnesampela privata*	Lepidoptera	Geometridae
'Cup moth'	*Doratifera quadriguttata*	Lepidoptera	Limacodidae
-	*Liparetrus* spp.	Coleoptera	Scarabaeidae
-	*Chrysophtharta amoena*	Coleoptera	Chrysomelidae
-	*Paropsisterna elliptica*	Coleoptera	Chrysomelidae
'Spitfires'	*Perga* spp.	Hymenoptera	Pergidae

The extent of infested forest was mapped by the Forests Department in 1964-1967 and in 1983, and by CSIRO in 1968-1976 and 1980. These maps, drawn to a common scale, are included in Abbott (1987).

The involvement of the Forests Department in leafminer research has until recently been minimal. Van Didden (1967) showed that an experimental aerial spraying of jarrah stands was ineffective and uneconomic. Since 1984 research has concentrated on measuring damage levels near Manjimup and Collie, relating these to environmental factors, and quantifying medium term impact on crown condition and wood increment (Abbott unpublished).

Information on silvicultural control and the cause(s) of the outbreak is less comprehensive and more speculative. The approach used in the CSIRO study involved sampling the abundance of larvae after they had ceased feeding, and expressing this variable in relation to a defined area of jarrah canopy. The abundance of moths emerging from soil has also been measured relative to a defined area of the forest floor. Studies in the same stands over several years have been used to assess the impact of prescribed low intensity fires, logging, clearing, and annual variation in rainfall on relative abundance of final instars and/or moths.

The approach used in the CALM research involved marking newly developing leaves each November (or as soon as the petiole is long enough to hold a tag) since 1984 in 20 stands within a 50-km radius of Manjimup. This research was extended in November 1987 to 20 stands within a 40-km radius of Collie. For each stand, damage caused to leaves by leafminer is expressed as a percentage of fully expanded leaf area. Multiple regression analyses were used to correlate this variable with environmental factors such as stand basal area, years since previous fire, years since previous logging, average annual rainfall, percentage forest cover and soil texture.

During 1983-1987 J. Hall of CALM estimated the extent of browning caused by leafminer in each jarrah crown in 15 plots north-east of Manjimup. Some of these trees experienced crown scorch during autumn burning during the study period.

Life history (Table 2)

Moths emerge and fly during the day. They are weak fliers. At night or during cold wet days they seek shelter in the litter. Almost all eggs are laid on the lower surface of jarrah leaves (Wallace 1970), at 14-22°C in diminishing light or overcast conditions (Mazanec 1986). The longevity of moths averages 10 days in the laboratory.

The caterpillar stage (within the leaf) lasts 160 days. After feeding finishes (instar 4), the caterpillar falls to the ground within a case, generally around midnight, to avoid predation by ants (Mazanec 1980). It then burrows 1-2 cm

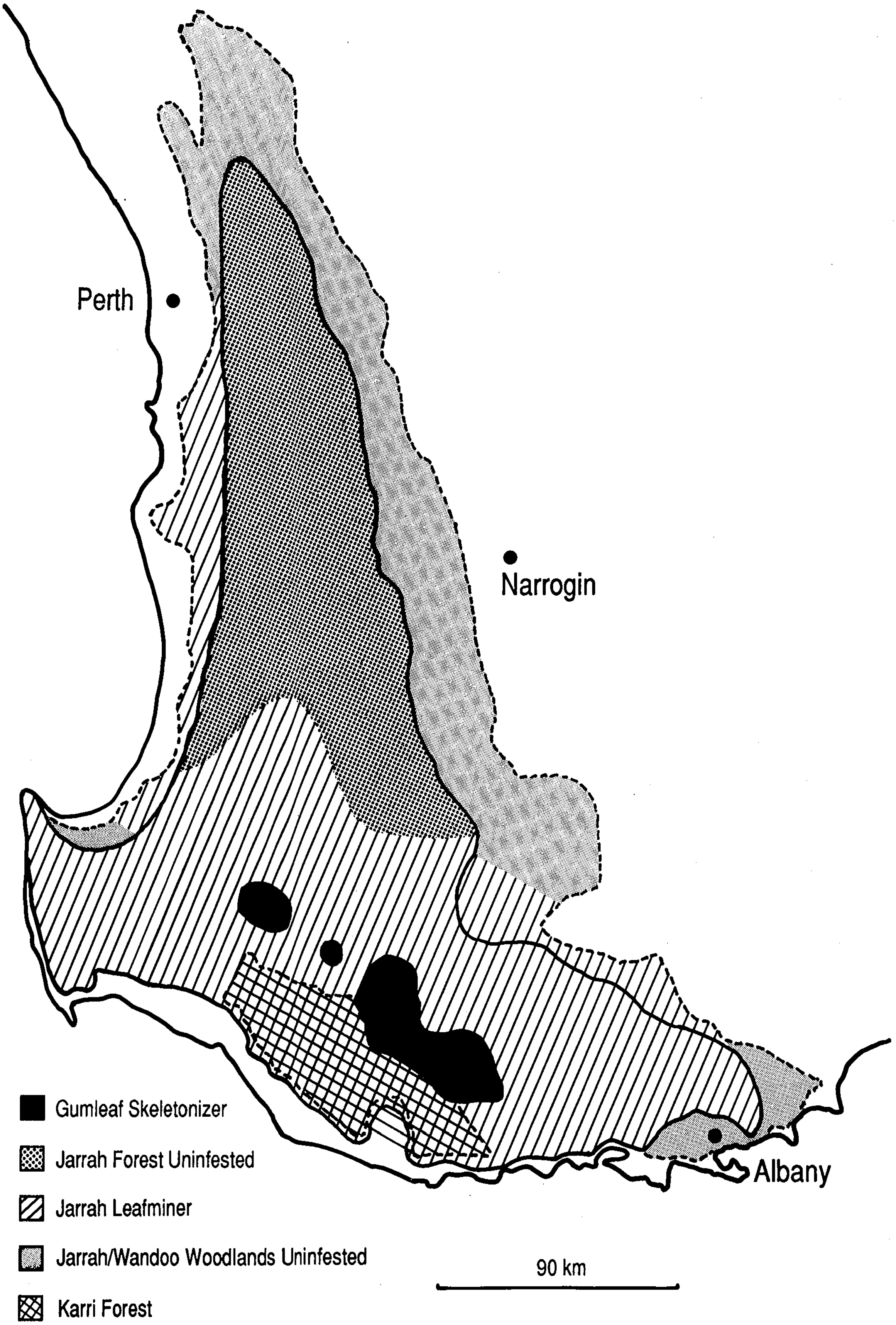

Figure 1.

Maximal extent of recent outbreaks of leafminer and skeletonizer in jarrah forest and woodland. There has been no forestwide survey of leafminer outbreaks since 1983. Since 1989, gumleaf skeletonizer has not been in outbreak.

into the soil, where it remains in diapause for about 150 days (depending on latitude), until the end of February, when pupation begins. The prepupal stage lasts 2 days and the pupal stage another 35 days.

Most eggs are laid in leaves 6-months-old (Wallace 1970; Mazanec 1986; Abbott unpublished) but there is conflicting evidence whether females prefer crowns of trees or foliage of ground coppice. Newman and Clark (1925) noted heaviest infestation nearer the ground and this was confirmed by Wallace (1970), with the exception of one stand lacking abundant advance growth. There infestation in the crowns 12 m above ground was extremely heavy. Mazanec and Justin (1986) found a statistically significant difference between infestation in upper canopy and lower canopy. CALM studies in 12 stands near Manjimup have not detected any significant difference in infestation 12 m above ground compared with 1-2 m above ground (Abbott unpublished). Nevertheless, where female moths have a choice, lower levels are used for oviposition first (Mazanec[1], personal communication).

Fire

Prescribed low intensity fires in spring, depending on their intensity, may kill most of the larvae diapausing in soil. But adults (moths) can disperse at least 1 km and probably farther passively. The result is that burnt stands are recolonized by moths the following autumn (Mazanec 1980, 1981). For up to 2 years following this type of fire, many more eggs are deposited than in adjacent unburnt stands.

Spring burning rejuvenates the forest crown, resulting in production of more leaves on which females prefer to oviposit the following autumn. These females then have a wider choice of leaves. Damage to individual leaves may not attain the 40 per cent threshold which the human eye perceives as brownish tinge in the forest canopy (Abbott, personal observation). The presence of abundant new foliage thus dilutes the effect of leafminer infestation.

Regression analyses of damage to tagged leaves (ground coppice) at 40 locations in the central and southern jarrah forest found 26 per cent of variation in damage was accounted for by period since fire (negative) and 14 per cent by average annual rainfall (also negative). Perhaps the latter correlation simply reflects the concentration of oviposition sites on few new leaves in stands in the low rainfall zone. Surveys of stands recently burnt in spring and adjacent stands not burnt for up to 11 years indicate no significant increase of leafminer density in the former (Abbott, unpublished).

Crown scorch during autumn fire caused a notable decrease in leafminer infestation (percentage damage) the following spring (Table 3). However, by the next spring, damage levels could not be distinguished from adjacent trees with unscorched crowns.

Drought

Drought (particularly during winter) results in decreased production of new leaves in the following spring and summer (Mazanec 1980, 1981). The number of eggs laid in the following autumn is therefore low, and so the leaves available should receive more oviposition, resulting in more damage and therefore obvious browning of the canopy.

Logging and regeneration

Logging followed by regeneration or thinning of stands results in rejuvenation of remaining foliage, perhaps responding to increased availability of soil moisture (Mazanec 1980, 1981); this favours oviposition. Advance growth is released and continues to grow and provide plentiful foliage suitable for oviposition. Detailed behavioural studies (Mazanec and Justin 1986) indicate that logging by itself should disfavour oviposition, because gravid females require subdued light and cool weather (<20°C). The abundance of new leaves (i.e. those with suitable oviposition sites) may be the true limiting factor.

Wallace (1970), using broad-scale surveys, concluded that severe infestations were usually associated with open country, or with natural or artificial clearings. He then suggested that logging and other silvicultural practices which tend to open up the forest may require modification. Mazanec (1980) examined this hypothesis more closely by measuring stand basal area at 42 places within the leafminer outbreak zone and at 38 places in non-outbreak areas in November 1978. The averages were 30 and 38 m^2ha^{-1} respectively, a statistically significant difference. However, recent research by CALM near Manjimup has shown that neither stand basal area nor time since logging are significant predictors of leaf damage by leafminer, but that rainfall zone is significant. These discrepancies can be reconciled because the difference between outbreak and non-outbreak areas in stand basal area very likely describes the difference between stands experiencing low or high average annual rainfall. Recent surveys of stands recently logged and adjacent stands either never logged or logged up to 100 years ago indicate no significant increase in leafminer density in the former (Abbott, unpublished).

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[1] Z. Mazanec, CSIRO Division of Entomology, Floreat W.A.

Table 2

LIFE CYCLES OF THREE PEST FOREST INSECTS

JARRAH LEAFMINER

	J	**F**	**M**	**A**	**M**	**J**	**J**	**A**	**S**	**O**	**N**	**D**
adult				X	X							
egg				X	X							
{instar 1					X	X						
{instar 2						X	X					
caterpillar {instar 3							X	X				
{instar 4							X	X	X	X		
larva in case in soil	X	X							X	X	X	X
prepupa		X	X									
pupa		X	X									

GUMLEAF SKELETONIZER

	J	**F**	**M**	**A**	**M**	**J**	**J**	**A**	**S**	**O**	**N**	**D**
adult		X	X									
egg		X	X	X	X	X						
{instar 1 (G)				X	X	X						
{instar 2 (G)					X	X	X					
{instar 3 (G)						X	X	X				
{instar 4 (G)							X	X	X			
{instar 5 (G)								X	X	X		
caterpillar {instar 6 (S/G,1head capsule)						X	X	X				
{instar 7 (S/G,2head capsules)							X	X	X			
{instar 8 (S/G,3head capsules)								X	X	X		
{instar 9 (S,4head capsules)	X	X								X	X	
{instar 10 (S,5head capsules)	X	X										X
{instar 11 (S,6head capsules)	X	X										X
pupa	X	X										

S = solitary phase

G = gregarious phase

S/G = some caterpillars are solitary, others are gregarious.

BULLSEYE BORER

	J	**F**	**M**	**A**	**M**	**J**	**J**	**A**	**S**	**O**	**N**	**D**
adult	X	X										X
egg	X	X										X
larva (instars unstudied) 2 years }	X	X	X	X	X	X	X	X	X	X	X	X
}	X	X	X	X	X	X	X	X	X			
pupa										X	X	

Table 3
EFFECT OF CROWN SCORCH DURING AUTUMN FIRES ON SUBSEQUENT INFESTATION OF LEAFMINER IN JARRAH CROWNS.

Plot No.	*Recent fire history*	*No trees*	*% of jarrah crowns infested[a] with leafminer in Spring*				
			1983	*1984*	*1985*	*1986*	*1987*
501	Spring 1980	15	18.7	21.0	16.7	35.0	17.3
505	Autumn1983	48	22.7	27.5	38.8	68.1	40.6
506	Autumn 1985[b]	31	32.9	27.3	22.4	26.3	22.6
557	Spring 1984	17	86.8	65.3	43.5	36.2	37.1
558	Spring 1984	15	95.0	76.7	65.7	62.3	60.8
559	Autumn 1986	16	93.8	92.5	69.4	53.1	82.2
		not scorched				*69.2*	*87.5*
		scorched				*2.5*	*66.3*
560	Spring 1984	20	92.5	59.8	43.0	50.3	59.0
561	Autumn 1986	25	88.0	64.6	51.6	6.0	11.6
		not scorched				*15.5*	*14.0*
		scorched				*0.0*	*11.1*
562	Spring 1981	27	55.0	52.8	57.2	42.4	42.8
564	Spring 1984	12	87.5	70.0	64.6	70.8	68.8
565	Autumn 1986	36	76.7	74.6	73.5	71.0	64.2
		not scorched				*75.2*	*68.8*
		scorched				*50.0*	*40.8*
566	Autumn 1986[b]	27	94.4	60.4	60.7	54.8	61.9
567	Spring 1984	13	96.2	47.3	38.5	55.4	61.2
618	Spring 1983	14	94.6	43.6	63.9	66.8	66.1

[a] at least 20 per cent of crown brown, caused by leafminer

[b] no crowns were scorched by this fire

Outbreaks

The occurrence of outbreaks of leafminer may be modelled in three ways (Fig. 2; Ridsdill-Smith[2] personal communication). Model 1 may apply to stands near Manjimup (infested for more than 20 years) whereas model 3 certainly would apply to Collie region (infested for 5-10 years). Model 2 is not supported by CSIRO data.

Reasons for a non-outbreak stand being transformed into an outbreak stand are not clear. Single factor explanations are not supported by the history of human use and management of the forest. Thus, fire was used by Aborigines; many areas near Collie and Manjimup were cut-over in the 1920s, yet these were free of outbreaks until recently. The outbreaks beginning east of Manjimup were recorded around farms near Lake Muir in the 1870s (Mazanec, personal communication). Yet the surrounding jarrah forest remained free of outbreaks until the 1960s.

Severe leafminer infestation itself promotes further infestation, because severely damaged leaves are shed in October instead of the normal January-February period (Mazanec 1980). This premature abscission contributes to an abundance of new leaves suitable for oviposition the following autumn.

Site

It is commonly observed that sandy sites in the southern jarrah forest consistently have browner canopies than upland sites. Sandy sites may result in more stress than

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[2] J. Ridsdill-Smith, CSIRO Division of Entomology, Floreat W.A.

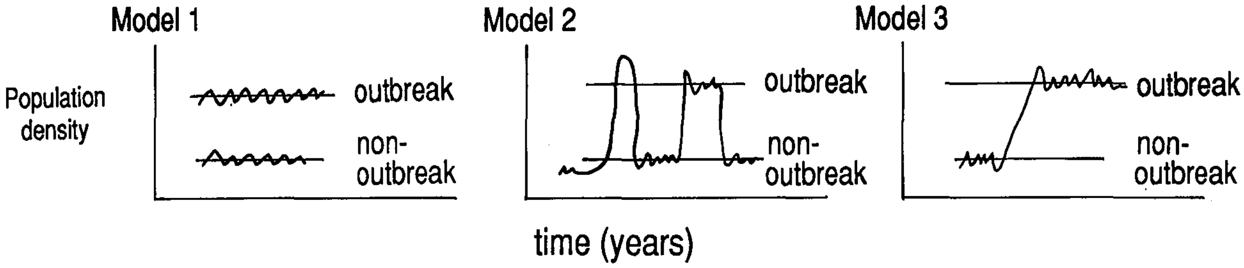

Figure 2.
Three models of how outbreaks of pest insect species may occur

Model 1 suggests that population density increases to outbreak levels then fluctuates somewhat about a mean density but not falling to non-outbreak levels over considerable periods of time (e.g. the Greenhouse Whitefly, F.D. Morgan, personal communication).

Model 2 shows typical periods of outbreak and non-outbreak varying in intensity and duration (e.g. Ips bark beetles, Sirex woodwasp).

Model 3 is in effect the early stage of Model 1.

lateritic sites, though the occurrence of the former in valleys contradicts this. Outbreaks were recorded in woodland (not forest) near Perth, Bunbury and Albany over 70 years ago (Newman and Clark 1925) - these are sandy sites.

One of the major obvious differences between the northern and southern jarrah forests is the more extensive gravelly soils in the former. The absence of leafminer from much of the northern jarrah forest may be related to this textural difference, given that the insect pupates in the soil. However, an experiment in the northern jarrah forest showed that gravelly surface soils are no impediment to larval penetration (Mazanec 1980).

Resistance

Not all jarrah are infested by leafminer to the same degree. Wallace (1970) termed individuals with only slight damage 'resistant' and those severely damaged 'susceptible'. Within outbreak areas, the crowns of resistant trees appear green whereas those of susceptible trees are brown. At low levels of infestation, the two types may be best distinguished (Mazanec 1980) by the abundance of aborted mines (containing dead larvae) and the scarcity of cutouts (made by fully developed larvae after they exit from their mines) within leaves of resistant jarrah.

CSIRO studies indicate that about 25 per cent of jarrah trees are resistant to larval feeding by leafminer (Mazanec 1974, 1980). One tree is known to be resistant to egg development and another tree is known to resist both oviposition and egg development (Mazanec 1985). This last tree differs from all other trees studied in its leaf anatomy; it also lacks tannin in its palisade tissue.

Analysis of data collected by CALM Inventory Branch suggests that none of the 406 jarrah trees assessed was strictly resistant to leafminer (Table 4). Yet Mazanec (1974) found that *c*.25 per cent of jarrah were 'resistant' to leafminer. This discrepancy can be reconciled by inferring that Mazanec tolerated the presence of some cutouts in crowns when he classified trees as resistant. These data imply that infestation by leafminer of tree crowns up to 20 per cent was the criterion subconsciously applied in the CSIRO study (Table 4). This yields an overall proportion of 'resistant trees' in these plots of 0.22, close to the figure quoted by Mazanec (1974, 1980).

Using this criterion, there is great variation among stands north-east of Manjimup in the proportion of jarrah 'resistant' to leafminer infestation (Table 5). Most stands had considerably less than 10 per cent of trees resistant. These plots also show that very few of the supposed resistant trees were in fact resistant each year between 1983 and 1987. This again highlights the incorrectness of equating 'absence of leafminer infestation' with 'resistance to leafminer infestation'.

Table 4
VARIOUS CRITERIA FOR RECOGNIZING JARRAH RESISTANT TO LEAFMINER INFESTATION IN CALM INVENTORY PLOTS IN THE SOUTHERN JARRAH FOREST

Proportion of jarrah with	*1983*	*'84*	*Spring* *'85*	*'86*	*'87*	*Grand Mean*
0 % leafminer infestation [a]	0	0	0.003	0.13	0.003	0.001
≤10 % leafminer infestation	0.09	0.03	0.05	0.19	0.21	0.11
≤20 % leafminer infestation	0.15	0.16	0.15	0.30	0.32	0.22
≤30 % leafminer infestation	0.27	0.33	0.29	0.44	0.44	0.35
≤40 % leafminer infestation	0.27	0.47	0.43	0.53	0.53	0.45
Total No. trees	323	373	375	375	374	

[a] Recognized as browning of jarrah crowns.

Table 5
SPATIAL VARIATION IN PROPORTION OF JARRAH 'RESISTANT' IN SPRING 1985 TO LEAFMINER INFESTATION Assuming ≤20 per cent infestation of each crown by leafminer as the criterion of 'resistance'

Plot No.	*Total No. trees*	*No. trees 'resistant'*
501	16	12
505	49	9
506	33	24
557	17	3
558	15	0
559	17	0
560	20	2
561	68	4
562	28	0
564	16	0
565	37	1
566	29	0
567	14	1
618	16	1
Total	375	57

Natural enemies

Mazanec (1987) recorded 10 species of Hymenoptera as parasitoids of the immature stages of leafminer. Parasitoids were more common in one plot within the northern jarrah forest (non-outbreak) compared with two plots in southern jarrah forest (outbreak). Relative mortality of feeding larvae was greater in non-outbreak than in outbreak situations.

Much is known (Mazanec 1988a, b, 1990a, b) about the basic biology of these parasitoids. Most species fluctuate markedly in numbers from year to year. A key factor analysis (Mazanec unpublished) indicates that these parasitoids are unlikely to prevent outbreaks. Most species have several generations per year and so require an alternative host during October to April when leafminer is within the soil. This is not available, so most of the parasitoids could never attain consistently high densities from year to year.

The two univoltine species are *Diadegma* sp. and *Chrysonotomyia* sp. C. The former is very rare but is not specific to leafminer, whereas the latter can be abundant, but is specific to leafminer (Mazanec, personal communication). Both species are internal parasitoids of larvae.

The most promising species, *Chrysonotomyia* sp. C, is apparently adversely affected by fire. Mazanec (1988a) found it present at high densities in the Meribup fire exclusion area (last burnt in 1962) in comparison with two stands experiencing the normal prescribed fire regime. Percentage parasitization by this species varies directly over the range 23-48 per cent with both annual variation in leafminer abundance and air temperature.

A systematic survey by CALM research staff in October 1987 of jarrah stands between Collie and Dwellingup found many examples of jarrah leaves with aborted mines, resulting from successful parasitoid control north of the outbreak. However, when the population density of leafminer increases as a result of increased oviposition, often helped by immigrating female moths, there is usually no proportional increase in parasitization (Mazanec 1988b). Birds also have minor impact because none irrupt during outbreaks and the local populations soon become satiated with leafminer larvae (Mazanec 1988b).

Key factor analysis

Mazanec (personal communication) has found that the main k factor in the life table of jarrah leafminer is natality. Once leafminer is present, the major cause of year to year variation in density is change in the abundance of leaves younger than six months.

Gumleaf skeletonizer

Since 1983 information has been gathered by the Forests Department and CALM on distribution of moderately and severely defoliated stands (1983, '84, '85 and 86), life cycle and hosts (Strelein 1988a), and damage levels in jarrah crowns and ground coppice, and development of a suitable method for sampling caterpillars (Abbott unpublished). This sampling has served to document the decline in skeletonizer density since 1987 (Farr[3], personal communication).

In the bulk of the forest there is only one generation each year (Table 2). The moth is nocturnal, does not feed, is a poor flier, and is probably shortlived. It mates in February and March and the female then lays her eggs in a series of parallel rows, usually on the underside of a leaf 1-4 months old. The eggs hatch from April to June and the small caterpillars feed together on the same leaf until about August. This gregarious phase lasts until about October when the caterpillars become largely solitary (2 or 3 per leaf) and more mobile, often using silken threads to cross to the other trees.

Caterpillars in the gregarious phase confine their feeding to the green leaf matter of the leaf, leaving patches of the brown network of veins. Those in the solitary phase are more voracious: they also eat smaller veins and frequently leave only the midrib and irregular portions of the lamina.

In stands of forest where caterpillars are dense, browning of crowns begins in late November to late December (depending probably on weather). In any case crowns are at their thinnest by the end of January when nearly all caterpillars pupate. Pupation occurs in cocoons spun by the caterpillars; these cocoons are placed on the branchlets near the leaves the caterpillars have been feeding on, in branch forks, or under bark flakes on large branches or the upper trunk, or in leaf litter.

Between January 1986 and January 1988, foliage from crowns of 45 jarrah poles, piles or trees (within a 50-km radius of Manjimup) was collected every 3 months and sorted for all individuals of invertebrate species present (Abbott, unpublished). The immediate environment of these 45 jarrah was quantified, as follows: basal area, proportion of jarrah present, years since previous fire, years since logging and average annual rainfall. The number of caterpillars expressed in terms of oven-dry leaf weight varied directly with rainfall. Thus, in contrast to leafminer, skeletonizer is more abundant in jarrah stands receiving higher rainfall. No significant statistical association between type of forest management and degree of infestation was found (Abbott unpublished).

Unlike leafminer, skeletonizer can feed on many eucalypt species, including karri, marri (*E. callophylla*), and wandoo (*E. wandoo*).

Strelein (1988a) and Abbott (1990) have circumstantial evidence that the extent of skeletonizer outbreaks since 1983 is greatest following dry, warm winters. However, it is not known whether this could be caused by annual variations in rainfall and minimum temperature acting directly on skeletonizer caterpillars or indirectly through parasitoids or predators or even the host tree.

Very few skeletonizer caterpillars occur on ground coppice (i.e. jarrah no taller than 1-2 m) (Abbott unpublished).

No jarrah poles, piles or trees resistant to skeletonizer feeding have yet been found. Even jarrah resistant to leafminer is susceptible to feeding by skeletonizer. During the past outbreak there was an urgent need for forest managers to arrange systematic searches for jarrah resistant to skeletonizer feeding. These searches are best carried out during January within skeletonizer outbreaks so that jarrah with green crowns would be obvious.

The only previously recorded outbreak of skeletonizer occurred in spring 1947 between Calingiri and Cowaramup. So few details are now available that it is uncertain whether the northern jarrah forest was affected by this outbreak.

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The most recent outbreak in the southern jarrah forest began in 1982, resulting in widespread browning of crowns in January 1983. Some 90 000 ha of jarrah forest between Nannup, Greenbushes and Walpole were affected. One year later the infested area covered 230 000 ha. In January 1985 and 1986 the extent of outbreak was 300 000 ha and 240 000 ha respectively. Maps showing the maximum annual extent of the outbreak (in January) have been produced for 1983, 1984, 1985 and 1986 (see Abbott 1987). The outbreak has since declined (Abbott 1990).

In January 1986 the severity of infestation by skeletonizer was quantified by crownometer in two stands near Yornup, based on 200 random observations in each stand. Crown cover was 66 per cent in the outbreak and 54 per cent in the non-outbreak stand. The proportion of brown jarrah foliage was 60 per cent and 2 per cent respectively, and that of brown marri foliage was 7 per cent and 0 per cent.

Bullseye borer

The life history (Table 2) of this insect was elucidated by Clark (1925) in a study done mainly on marri near Mundaring. Interestingly, although recording six eucalypt species as hosts, Clark did not include karri. Voutier (unpublished) studied aspects of the biology of this species in karri from 1983 to 1985, and found several differences from Clark's study, particularly the presence of 'frass ejection vents' along the bole, where the larva empties out the contents of its tunnel. Infestation was found to be widespread throughout the karri forest and have the capacity to inflict severe structural damage to young vigorous karri regeneration.

Borer damage is also associated with brown wood, i.e. heartwood, which develops rapidly into decayed wood (Donnelly, unpublished), and with kino exudation from frass ejection vents (Voutier, unpublished). Interestingly, Clark (1925) vigorously disputed that this borer was the main cause of kino veins in marri.

In 1986 R. Smith (CALM Research Division) studied the association between internal borer damage and external symptoms on karri trees, and statistical analysis showed that the association was significant. These data revealed that infestation of karri regeneration began sometime between 10 and 14 years of age. In 1987, Smith surveyed 30 stands of karri regeneration, and recorded on a sample of 50 trees in each stand five indications of external damage caused by bullseye borer, dominance status, bole length, stem diameter and bark thickness at breast height. Stand age, years since the last fire, coupe size, site type, average annual rainfall, stand basal area, proportion of marri present and distance to nearest non-regenerated karri forest were also recorded for the stand.

Most bullseye borer damage is associated with proximity to mature karri stands, sites where karri is a minor component before regeneration, small coupe size and site susceptibility to drought (Abbott *et al.* in press).

ECOLOGICAL IMPLICATIONS

The impact of pest insect species on the ecology of jarrah and karri forest has been measured for only a few attributes, namely biomass of other crown-dwelling invertebrates, condition of jarrah crowns, wood increment, and mortality of jarrah.

It was fortunate that leafminer outbreaks developed in W.A. decades after methods of organic chemical control became commercially available in the 1940s and had been enthusiastically adopted in North America and Europe to control insect outbreaks. By the 1960s it was widely appreciated that insecticides have adverse environmental and evolutionary consequences, including water pollution, contamination of birds and mammals feeding on affected insects, extensive mortality of non-target insects, and evolution of resistance by pest insects to chemicals, which counteracts their efficacy. Insecticides have not been used to reduce leafminer outbreaks.

Conservation values

It is reasonable to hypothesize that stands with insect-damaged crowns would offer fewer feeding sites, less food and fewer suitable oviposition sites for other invertebrates compared with stands in which most of the foliage comprising the canopy is green.

In the southern jarrah forest, samples have been collected (9 sampling dates over 2 years x 45 trees) from jarrah both unaffected and affected to varying degrees by gumleaf skeletonizer caterpillars. These samples have been sorted to species level, and the biomass of each species has been estimated from a body length/oven-dry mass regression based on a representative sample of species. Biomass of each sample is expressed relative to oven-dry mass of foliage in the sample.

A graph (Fig. 3) showing the biomass of all invertebrates (excepting skeletonizer) in relation to the biomass of skeletonizer caterpillars in the same sample clearly shows that there is no significant relationship. Similar graphs were prepared for Coleoptera, Araneae, Hemiptera, other Lepidoptra, Hymenoptera, Diptera, other leaf chewers,

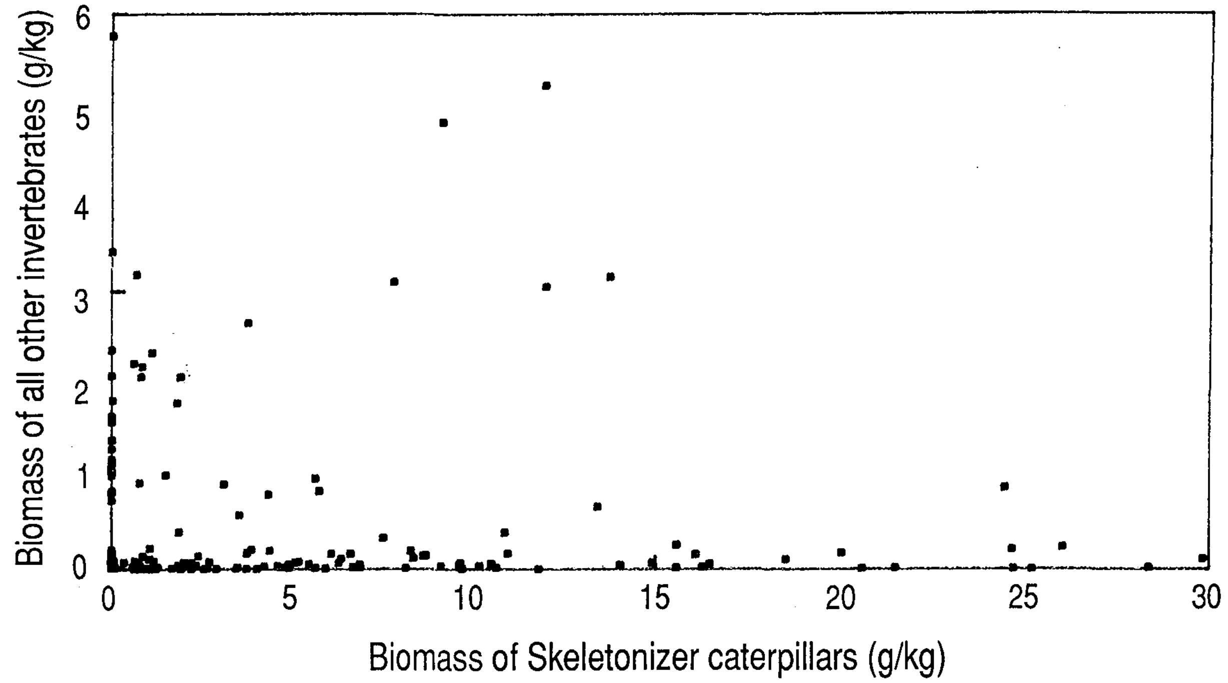

Figure 3.

Biomass of all invertebrates (kilogram oven-dried/kilogram foliage oven-dried) graphed against biomass of skeletonizer caterpillars (same units) in samples collected from crowns of jarrah poles in the southern jarrah forest during 1986-1988 (N=405).

parasitoids, and predators. None showed any significant correlation. Therefore, skeletonizer has not caused any measurable deterioration in the microhabitat of the hundreds of other indigenous/endemic invertebrate species living in jarrah crowns.

Stand dynamics and productivity

Assessment of whether jarrah crowns are deteriorating in width, depth and leaf density

In 1970 a questionnaire (M. Wallace, CSIRO, unpublished) asked foresters at Manjimup whether leafminer attack affected the forest canopy and general tree vigour in any viable way and whether any visible change was confined to jarrah leafminer infested areas only.

Most foresters circularized believed that permanent damage resulted from jarrah leafminer infestation. They noted that where defoliation had only recently begun, crown deterioration had not reached the severity of old infestations. Deterioration appeared rapidly within 5-10 years. Photographs of selected jarrah crowns (Fig. 4) best show the extent of this deterioration.

These conclusions are supported by studies on inventory plots rating the condition of the crown of selected codominant and subdominant jarrah resistant or susceptible to leafminer. C. Ward (CALM Research Division) used a modified version of an Index (Grimes 1978) based on visual estimates of leaf density, contribution of epicormic branches to foliage and the incidence of dead branches. The index ranges from 0 (tree dead) to 24 (see Appendix).

Resistant poles in high and low quality jarrah forest had above average (=12) indices of crown condition (Table 6). The deterioration in crown condition caused by leafminer was greatest in codominant poles in low quality forest (44 per cent), followed next by codominant poles in high quality forest (35 per cent). Crowns of subdominant poles deteriorated least (19 per cent). These data, when combined with those in Table 7, indicate that diameter under bark (d.u.b.) increment would be zero when the crown condition index has a value of 9-10 or below.

CALM policy concerning thinning of jarrah stands is that all resistant trees are retained regardless of size, position or quality (Bradshaw 1987).

The conditions of the crowns of tagged jarrah poles was assessed in 1988 on plots established by Mazanec (personal

Figure 4a Crown of healthy mature jarrah pole. Crown condition index = 23

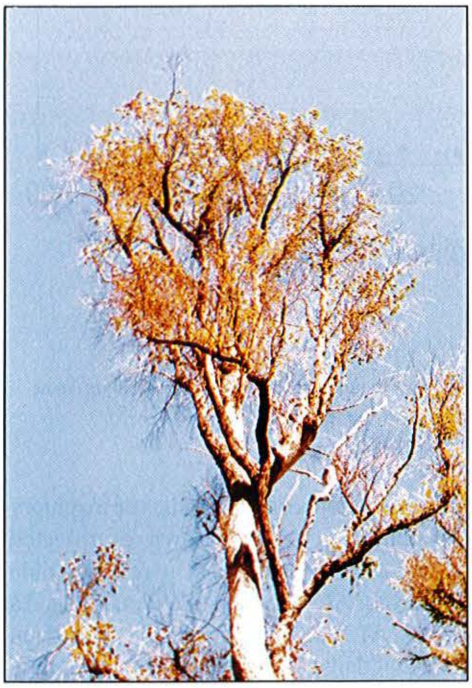

Figure 4b Crown suffering from recent insect infestation. Crown condition index = 14

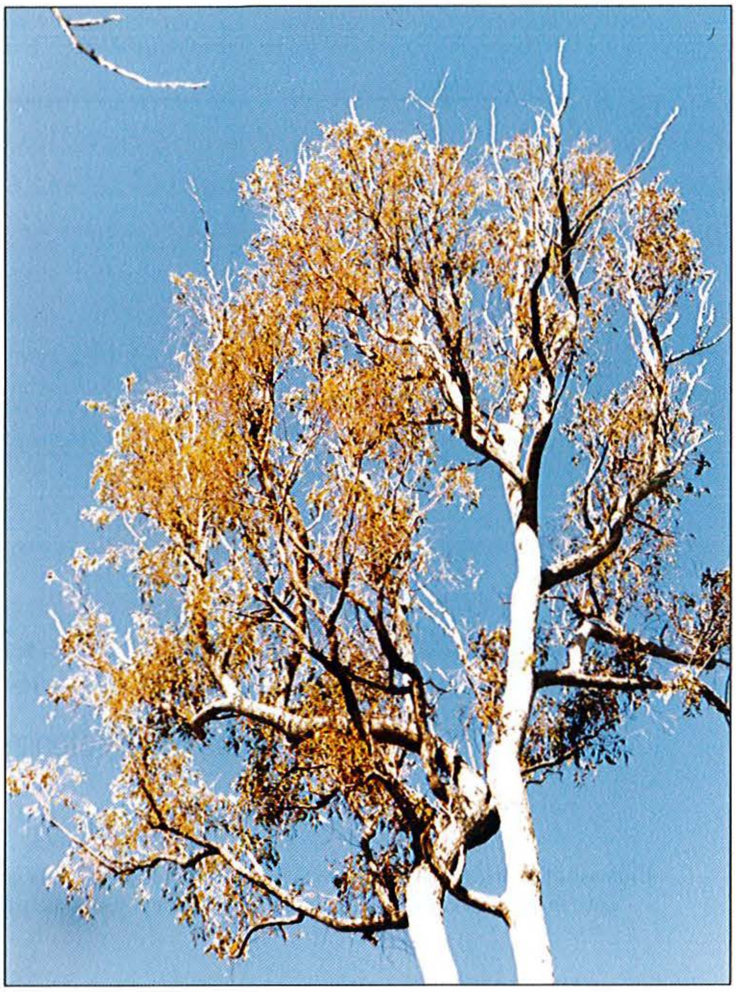

Figure 4c Crown showing early signs of degradation. Crown condition index = 13

Figure 4d Crown showing severe degradation. Crown condition index = 10

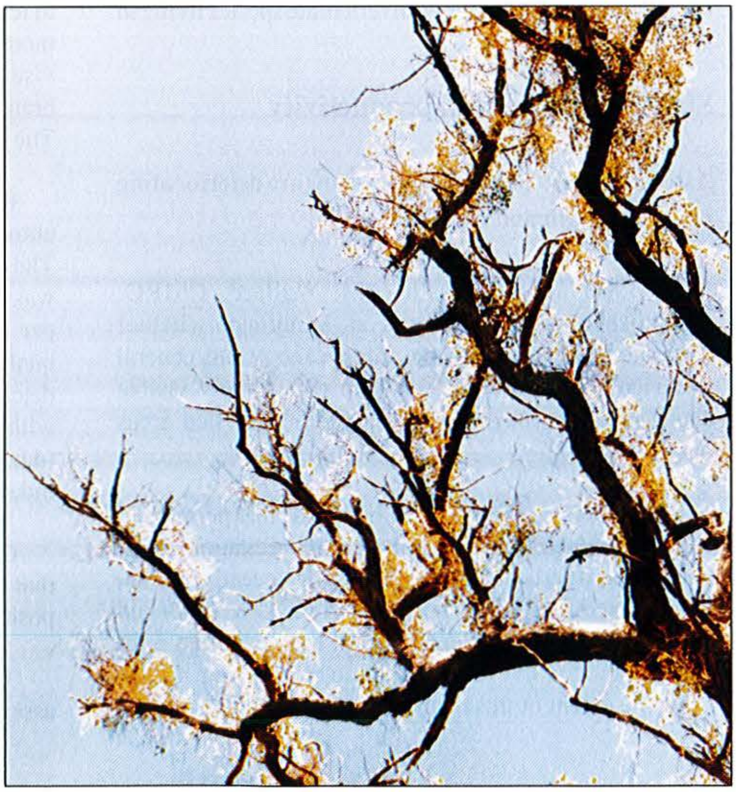

Figure 4e Detail of crown showing severe degradation. Crown condition = 12

Figure 4f Full extent of pest insect impact on jarrah crowns is obvious when seen from afar. High quality forest

Figure 4g Full extent of pest insect impact on jarrah crowns is obvious when seen from afar. Low quality forest

Table 6
MEAN INDEX OF CROWN CONDITION FOR CODOMINANT AND SUBDOMINANT JARRAH POLES IN HIGH AND LOW QUALITY STANDS IN THE SOUTHERN JARRAH FOREST

Stand quality	*Crown condition*			
	Codominant		*Subdominant*	
	Resistant	*Susceptible*	*Resistant*	*Susceptible*
High (9 plots)	17.5	11.3	15.6	12.7
Low (6 plots)	17.0	9.5	16.2	13.1

Table 7
MEAN D.U.B. INCREMENT (cm y^{-1}) SHOWN BY CODOMINANT AND SUBDOMINANT JARRAH POLES IN HIGH AND LOW QUALITY STANDS IN THE SOUTHERN JARRAH FOREST

Stand quality	*d.u.b. increment*			
	Codominant		*Subdominant*	
	Resistant	*Susceptible*	*Resistant*	*Susceptible*
High (9 plots)	0.23	0.11	0.19	0.15
Low (6 plots)	0.16	0.04	0.14	0.06

communication) in 1967. The two plots in high quality forest were free of leafminer outbreaks in 1968 whereas the two plots in low quality forest had experienced outbreaks for at least 5 years (Mazanec, personal communication). Poles were selected in 1968 on the basis of having a perfect crown (Mazanec, personal communication), equivalent to 24 in the rating scheme used in 1988 (Appendix).

Resistant trees (as designated by Mazanec) in all plots showed remarkably similar declines in crown index (-24.2 to -29.6 per cent, average -26.8 per cent) over the 21-year period (Table 8). This presumably is caused mainly by drought. Fire was not a factor as one plot had experienced no fire since 1962 (Table 8).

Susceptible trees (as designated by Mazanec) showed an average crown decline of -40.5 per cent (range -30.4 to -46.3 per cent) during the 21-year period (Table 8). Comparison of the crown condition of susceptible and resistant poles in each plot shows that the average difference is 18.2 per cent (range 8.2 - 28.3 per cent), i.e. leafminer outbreaks over two decades are responsible for nearly 20 per cent of the deterioration in jarrah crown condition.

Outbreaks of leafminer reached Collie district between 1975 and 1983. A comparison in 1987 of jarrah plots within the outbreak zone, with those to the north of the outbreak shows that poles in high quality forest experienced most deterioration in crown condition (Table 9). Poles in low quality forest had similar crown condition irrespective of location with respect to outbreaks, perhaps a legacy of past wildfires. Similar relationships were found for diameter under bark (d.u.b.) increment (Table 9).

Table 8
MEAN INDEX OF CROWN CONDITION IN 1988 FOR JARRAH POLES IN HIGH AND LOW QUALITY STANDS IN THE SOUTHERN JARRAH FOREST. ALL TREES IN 1967 HAD INDEX VALUE = 24.

Stand quality	*Crown condition*	
	Resistant	*Susceptible*
High		
Diamond	18.2 (13)	16.7 (12)
Marranup	16.9 (14)	14.1 (15)
Low		
Boyicup	18.0 (2)	12.9 (9)
Cessna	17.2 (10)	13.8 (10)

NOTE: (1) All but 6 of the 85 trees assessed in 1988 had subdominant crowns.

(2) Year when last burnt : Diamond (Spring 1973), Marranup (Spring 1985), Boyicup (Autumn 1981), Cessna (Spring 1962).

Table 9
MEAN INDEX OF CROWN CONDITION AND MEAN D.U.B. INCREMENT (cm y^{-1}) FOR CODOMINANT AND SUBDOMINANT JARRAH POLES IN HIGH AND LOW QUALITY STANDS IN COLLIE DISTRICT (NOVEMBER 1987).

All jarrah were classified as susceptible to leafminer.

Stand quality	*Codominant*		*Subdominant*	
	Crown index	*d.u.b. increment*	*Crown index*	*d.u.b. increment*
High				
- in outbreak zone				
(4 plots)	13.6	0.11	14.3	0.15
- not in outbreak zone				
(3 plots)	19.4	0.40	19.3	0.51
Low				
- in outbreak zone				
(3 plots)	14.2	0.11	16.7	0.07
- not in outbreak zone				
(3 plots)	14.2	0.07	17.4	0.16

NOTE: Outbreaks of leafminer reached these plots between 1975 and 1983.

Wood growth

It has long been recognized that leaf-eating insects reduce the growth rate of Australian forests (Jacobs 1955).

Short-term effect of leafminer on wood growth

Mazanec (1974) assessed the short-term impact of leafminer on diameter increment by measuring the diameter (over bark) of 58 resistant trees and an equal number of susceptible trees on four sites near Manjimup from 1967 to 1971.

One site, Diamond, had very low levels of leaf damage (not specified by Mazanec 1974, but certainly <40 per cent, Mazanec personal communication), and a comparison of the diameter increments of resistant and susceptible trees there showed no appreciable difference. However, the other three sites experienced 'severe' leaf damage (i.e. 60 per cent of the average leaf area of susceptible jarrah was damaged by leafminer, Mazanec personal communication), so that the annual loss of diameter increment per susceptible tree at these sites ranged from 64-79 per cent (mean 71 per cent). The annual loss of diameter increment per tree (susceptible and resistant jarrah combined) varied from 49-61 per cent (mean 54 per cent). Thus, within severely infested forest, diameter increment of the jarrah component is about half of its value in non-outbreak forest (Mazanec 1974).

Mazanec (1974) estimated that 400 000 ha (25 per cent) of State jarrah forest was affected by moderate to heavy leafminer infestation (i.e. an average of 40 per cent of leaf area damaged). The loss in wood growth over such a large area is of concern.

The four plots just mentioned were a subset of seven plots established by Mazanec in 1969 in which 20 trees were tagged. An additional four plots were established in 1980.

Medium and long-term effects of leafminer on wood growth

Medium-term effects have been assessed on a selection of inventory jarrah plots in Manjimup, Pemberton and Nannup Districts. Most of these plots were first measured in 1961 or 1971. C. Ward (CALM Research Division) visited 15 plots, selected codominant and subdominant jarrah poles, noted whether they were resistant or susceptible to leafminer and remeasured diameter over bark (d.o.b.) and bark thickness. These data allow calculation of growth losses over 15-25 year periods (Table 7).

The diameter (under bark) loss in high quality stands is 52 per cent for codominant poles and 21 per cent for subdominant poles (or an average of 38 per cent). In low quality stands the comparable figures are 75 per cent and 57 per cent (66 per cent). Because future intensive silviculture will be carried out only in high quality stands, the loss in low quality forest is not relevant. To put these data on a stand basis, it is necessary to multiply by 0.75 (the proportion of susceptible trees). Assuming that average volume under bark (u.b.) production is 1 m^3 ha^{-1} per year and given that 420 000 ha of high quality jarrah forest is infested by leafminer, then the *annual* loss in wood growth caused by leafminer is about 0.38 x 0.75 x 420 000 or 120 000 m^3. Allowing for 50 per cent utilization, this amounts to a loss of more than $1m in sawlog royalty each year. These figures could be made more precise if the volume of wood produced per hectare per year by codominant and subdominant jarrah in high quality stands in the southern jarrah forest were known.

Effect of skeletonizer on wood growth

About 118 000 ha of high quality jarrah forest was infested moderately-severely by skeletonizer at the latest estimate (January 1986). Given that the proportion of jarrah resistant to skeletonizer appears << 1 per cent, then by analogy with leafminer the loss of wood growth owing to skeletonizer should be 118 000 x 0.38 or 45 000 m^3 y^{-1}. This was equivalent to about $0.35m per annum in sawlog royalty, allowing for 50 per cent utilization.

Effect of bullseye borer on wood growth

The main impact of this borer is that its tunnels within the karri bole affect the utilization of timber. However, as yet, no detailed estimates of wood loss are available.

Mortality of jarrah

Effect of chronic infestation

Leafminer outbreaks have not yet resulted in any extraordinary mortality of jarrah trees, as evidenced from 11 plots kept under long-term investigation by CSIRO. In six plots (Cessna, Boyicup, Boyicup East, Marranup, Diamond and Dingup) established in 1969, only three out of 120 trees had died by 1987; and one of these trees had never experienced much leafminer damage (Mazanec, personal communication).

In four other plots (Gold Gully, Stallard, Easter and Jubilee) established in 1980, none of the 80 trees had died up to 1987.

Combining these data, mortality owing to leafminer is 0.11 per cent per annum, i.e. 11 trees die per 10 000 trees per annum.

Dynamic processes

The intermediate (900-1100 mm p.a.) and low (<900 mm p.a.) rainfall zones cover the parts of the jarrah forest most favoured by leafminer. The most relevant watercourses include the Helena, Canning, Serpentine, Harvey, Collie, and Donnelly Rivers and Bell Brook, Bingham River, Harris River, Beraking Brook, Darkin River, Nockine Brook, and Bannister River. Though several of these streams feed into the reservoirs serving metropolitan Perth, recent related research has indicated that any forest decline over the next 20 years should not cause appreciable rises in water table or increases in water salinity (Stokes and Batini 1986).

CURRENT RESEARCH

Details of current research activities are set out in CALM's Research Division Research Plan, 2nd - 4th editions. These activities are only summarized here.

Insect ecology

Leafminer

When leafminer completes its feeding in the jarrah leaf, it cuts itself out of the leaf while remaining in a cell, which falls to the ground. Since October 1986, the presence of 'cutouts' in leaves has been recorded at predetermined sampling points in forest north of the Collie outbreak. These data allow accurate measurement of the rate of spread of leafminer populations north of the outbreak.

The level of parasitism of leafminer larvae in mines within, and to the north of, the leafminer outbreak near Collie is being assessed.

The effect of moderate intensity fire on leafminer and condition of crowns is being assessed in 240 ha of jarrah forest north of Collie which was burnt in autumn 1989. It is hoped that these fires could be used to reduce the rate of spread of the outbreak farther into the northern jarrah forest.

The effect of leafminer damage on biomass of other invertebrates in jarrah crowns is being investigated at 15 sites in Collie district.

Skeletonizer

Studies of the population dynamics of caterpillars have commenced. These will indicate between which instars most mortality occurs.

Laboratory cultures of caterpillars collected from forest are being used to detect parasitoids. The preferred site of pupation is being studied as a possible means of population control.

The impact of an autumn fire on population densities is being assessed in an area of 100 ha north-west of Manjimup.

The effect of size of the female moth on fecundity is being measured.

The northern limit of distribution in jarrah forest has recently been determined.

The occurrence of bivoltine populations and their taxonomic status is under investigation.

Bullseye borer

Studies of oviposition preferences are underway.

Monitoring levels of herbivory in jarrah forest

In 1984, an unbiased method of sampling leaves was developed (Abbott 1987). A herbarium of damaged foliage, consisting of voucher specimens on which insects were observed feeding, has been compiled. It is now possible to reliably assign most examples of damaged foliage to a particular insect species or family.

Damage levels in ground coppice jarrah in Manjimup region have been quantified annually since 1984, and since 1987 in Collie region. Damage levels in pole crowns have been measured less frequently. Damage levels in ground coppice in the Jarrahdale-Dwellingup region were measured on the 1985, 1987 and 1989 cohorts of leaves.

Variation in damage levels in terms of site and season are now well understood. Multiple regression analyses are being used to relate damage levels to environmental factors. The complexity of interacting factors makes factorial experimentation impracticable at this stage.

Research into economic injury levels has focused on manual defoliation of ground coppice jarrah. It is reasonable to expect that a threshold exists, below which damage has no impact on jarrah growth. An experiment using ground coppice in Holmes Block was set up in December 1987 to assess the interaction between proportion of foliage removed (0,25,50,75,100 per cent) and number of annual defoliations (0,1,2,3).

Monitoring potential pests

More than 20 insect species have been found damaging jarrah, marri or karri leaves. Surveillance is maintained on several species which are known outside the forest or elsewhere in Australia to occasionally cause severe defoliation. These include *Phylacteophaga froggatti, Mnesampela privata, Doratifera quadriguttata, Liparetrus* spp, *Chrysophtharta amoena, Paropsisterna elliptica* and *Perga* spp.

P. froggatti (leafblister sawfly) was introduced into Western Australia in 1978 (Curry 1981) and rapidly spread throughout the south-west. Jarrah is not attacked but marri is very susceptible to defoliation. Curiously, although the species has been recorded at Boyanup and Manjimup, it has so far not penetrated into the forest.

The remaining species listed are indigenous. Several of them are widespread in jarrah forest but never locally abundant. Life history details are collected opportunistically for each of these species.

A species of leafminer has been recorded sporadically on leaves of karri in the main karri belt. In the Porongurup Range, this species has recently caused significant damage to the foliage of karri saplings.

Mapping of outbreak areas and hazard rating of stands

The current extent of the outbreak of leafminer favours the use of remote sensing techniques to map insect damage in forests. The CSIRO Remote Sensing Group is involved in determining the spectral, temporal and spatial characteristics required of a remote sensing system to enable discrimination of insect-infested forest (Behn *et al.* 1990). One test area (Collie) is being used; results to date are promising and may lead to a practical and routine inventory system being implemented.

Any surveys must be carried out in October (for leafminer) and in January (for skeletonizer) because they depend on the recognition of brown crowns. If deferred, surveys are useless because the flush of new green leaves after insects cease feeding quickly obscures insect damage.

Hazard rating is a recognition that stands differ in their vulnerability to damage by insects. Hazard is determined by the influence of climatic, site and management factors on observed damage levels. It differs from risk, which is an assessment of the probability of spread from nearby stands experiencing insect outbreaks or after a particular forest operation. Reliable prediction of hazard and risk would enable the forest manager to judge which forest management practices should aggravate outbreaks. The aim of forest management in terms of outbreaks is to know how stands free of outbreaks can be kept in this condition, and how stands experiencing outbreaks can be manipulated to reduce damage to an acceptable level. In State forests, the aim is to reduce infestation and so promote wood growth and perfect conservation and aesthetic values, whereas in national parks and reserves the goal is to improve aesthetics and nature conservation values.

CALM is planning to map the occurrence in selected areas of the site-vegetation types devised by Havel (1975) and Strelein (1988). These maps will then be used to assess whether particular site-vegetation types associate strongly with susceptibility to infestation by leafminer or skeletonizer. Such information would allow CALM to plan to manage certain site-vegetation types differently from others.

The likely future distribution of outbreaks of leafminer and skeletonizer is not clear. Both species occur at present well north of the current area affected by outbreaks (Fig. 1). North of Dwellingup, leafminer was recorded up to 1988 at low densities in Randall, Lesley, Flynn, and Victoria forest blocks, and since that year in many forest blocks. The only records of skeletonizer north of Collie are from Illawarra, Lesley, Randall, Holmes, Clark and Proprietary forest blocks. However, outside the forest, skeletonizer occurs very widely in an area bounded by the coastline, Geraldton, Merredin and Esperance (Abbott 1987).

If the rate of expansion of leafminer outbreaks follows that experienced during the 1960s and '70s near Manjimup, then the leafminer outbreak should reach Dwellingup by 2011, Jarrahdale by 2028 and Mundaring by 2043.

Monitoring of jarrah crowns

The condition of crowns of jarrah is being monitored regularly near Collie (180 trees) and near Manjimup (45 trees). This procedure should provide early warning of any marked deterioration in crown condition or unusual mortality resulting from outbreak populations of defoliating insects.

The current inventory of jarrah forest being carried out by Inventory Branch of CALM should provide forest-wide estimates of jarrah mortality. Stands with known histories of insect outbreaks could then be compared both with each other and with stands in non-outbreak zones.

HIGH PRIORITY ADDITIONAL RESEARCH REQUIREMENTS

1. *Impact of repeated, chronic defoliation on nutrient levels in, and physiological condition of, jarrah*

A physiologist could examine small numbers of trees, within and without outbreak areas, and resistant or susceptible to defoliators, and draw precise conclusions about impacts of defoliators on the well-being of jarrah trees. Very early warning of any impending mass collapse of stands would be gained. It would also be valuable to evaluate the physiological response of jarrah to a combination of experimental intensities and repetitions of defoliation.

2. *Reasons for the initiation and continuation of outbreaks*

The outbreak of leafminer that began more than two decades ago in the southern jarrah forest had no historical precedent, since European settlement. Newman and Clark (1925) recorded that noticeable damage was confined to the 'coastal jarrah growing on the plains country', mainly near Fremantle, Bunbury, Busselton and Albany, but radiating inland ' for several miles to the base of the foothills'. They noted explicitly that no damage was recorded in the country between Torbay and Busselton or in the Darling Range forests. Outbreaks near Perth (Kings Park) were first recorded in 1914.

This historical perspective invites the question 'What initiated the 1960s outbreak in forest east of Manjimup?' Obvious perturbating factors include fire (a change in intensity, season of burning, frequency of burning or combinations of all three factors), logging, deforestation (for farming and settlement) and subtle changes in climate (rainfall, temperature, soil dryness index). Mazanec (1988b) argues that any factor that leads to the production of new leaves will lead to an increase in population of leafminer.

Associations between some of these variables and expansion of leafminer and skeletonizer outbreaks could be sought by modelling. The maps in Abbott (1987) would serve as a start. Then the more accurate maps that would be produced using remote sensing techniques would allow refinement of these models. It is hoped that these models could explain why leafminer and skeletonizer have become forest pests only within the past 25 and 5 years respectively.

3. *Search for parasitoids of leafminer and skeletonizer elsewhere in Australia*

This would complement the research completed by CSIRO on local parasitoids of leafminer and the research being done by CALM on skeletonizer parasitoids already present in the southern jarrah forest. Research interstate would ideally be covered by the CSIRO Division of Entomology and the Department of Entomology, Waite Institute of Agricultural Science, South Australia.

4. *Virological, bacteriological and/or mycological options for controlling leafminer and skeletonizer*

If research on parasitoids yields little of value, it may be necessary to study the natural importance of viruses, bacteria and fungi in killing pest insects, and to evaluate whether other viruses, bacteria and fungi should be introduced to control pest insect populations without detriment to non-pest insect populations.

In particular there is a need to examine whether aerial spraying of foliage with commercially available *Bacillus thuringiensis* would contribute to control of leafminer, without prejudicing conservation values by reducing the abundance of other invertebrates (particularly Lepidoptera) living in jarrah crowns.

5. *Selection of jarrah resistant to leafminer and skeletonizer*

A workable definition of resistance needs to be devised. Trees and regeneration immune to pest insect infestation could then be identified and retained throughout the jarrah forest. The feasibility of readily increasing the proportion of resistant trees in stands requires investigation.

It is necessary to test trees with resistant phenotypes (i.e. crowns unaffected in stands severely affected) and establish a genetic basis by collecting seed, growing seedlings and testing them in culture and within forest.

6. *Comparison of bird populations within and outside outbreak areas*

Extensive browning of jarrah crowns may lessen their suitability as habitat for insectivorous birds. Expansion of outbreak areas may result in some of these bird species declining in distribution and abundance within jarrah forest. The suitability for birds inhabiting crowns of unaffected species such as marri or yarri also needs to be assessed.

7. *Taxonomic expertise*

The taxonomy of many potentially economically important insect groups is out-of-date or confused. The major taxa are Chrysomelidae, Cerambycidae, Curculionidae, *Doratifera*, *Lyctus* and *Ochrogaster*.

This would be an ideal subject for CSIRO to service, as the Division of Entomology has the major national collection (ANIC) and the largest national group of insect taxonomists.

8. *Nitrogen fertilization and thinning of jarrah stands*

There is a need to determine whether proposed operational thinning and application of nitrogen fertilizers (during winter) will favour or disfavour leaf-feeding by leafminer and other insect species.

9. *Damage to foliage of* Eucalyptus rudis

A limited comparison in 1985 of damage by insects, fungi and other agents to the leaves of the eight eucalypt species growing in forest in the Manjimup region (Abbott unpublished) found that *E. rudis* experienced almost as much damage as jarrah. To the east of State forest *E. rudis* becomes an important component of the lower parts of the landscape and may help mitigate salinity problems in agricultural areas in the lower south-west. There is a need to investigate why *E. rudis* is attractive to insect herbivores.

10. *Impact of outbreaks on water quality*

Historical records of groundwater levels, stream flow and salinity levels in subcatchments in the northern and southern forests should be examined to determine whether leafminer infestation has produced measurable changes. These subcatchments have or should have <900 mm annual rainfall, be free of any deforestation, lack wandoo and have comparable densities of marri. Hydrologic records for the period 1940-1960 (before leafminer outbreaks) could be compared with those for the period 1960-1980, during which leafminer outbreaks have persisted. Comparisons between adjacent subcatchments in the southern jarrah forest (one severely affected by leafminer, the other not so) would also be useful but difficult.

11. *Plantations of* Eucalyptus globulus

Plans to reforest tens of thousands of hectares of farmland in the south-west could result in insect outbreaks (especially *Mnesampela privata* and Psyllid species) developing in these stands and then spreading to adjacent native hardwood forests.

Monitoring of insect populations and experimental studies of chemical, silvicultural and biological control should be an essential component of the development of these plantations.

12. *Argentine Ant*

This species, *Iridomyrmex humilis*, was accidentally introduced to Western Australia in 1941. It is a very aggressive species, eliminating most native ant species. In South Africa, displacement of seed-dispersing native ant species resulted in reduced germination of many native plant species. There is therefore a need to survey the jarrah and karri forest for Argentine ants and if necessary carry out control measures.

CONCLUSION

Factors affecting the long-term population dynamics of jarrah leafminer and gumleaf skeletonizer are poorly understood. The evidence available indicates that forest operations have not been responsible for recent outbreaks of these insects. It is not yet possible to assess the problem of bullseye borer outbreaks because old-growth karri trees have not been sampled.

From an economic perspective, insect problems in jarrah forest are probably as serious as dieback in jarrah forest caused by the fungus *Phytophthora cinnamomi.*

If pest insect outbreaks can be controlled, then the following benefits would follow: improvement in wood productivity; improvement in the condition of tree crowns, lessening the risk of death of jarrah after long periods of intense defoliation, and consequential detrimental effects on forest hydrology, conservation, and aesthetics.

The hardwood forests of south-western Australia are currently valued in excess of $1 000m (G. Malajczuk[4] personal communication). They are therefore a significant asset deserving of protection from outbreaks of injurious insects.

ACKNOWLEDGEMENTS

I thank W. Boardman, T. Burbidge, D. Donnelly, J. Farr, J. Hall, Z. Mazanec, J. Ridsdill-Smith, R. Smith, G. Strelein, P. Van Heurck, R. Voutier, C. Ward and M. Williams for providing information used in this report, and G. Godfrey for word-processing.

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APPENDIX

Assessment of jarrah crowns (based on method of Grimes 1978). The value of the index ranges from 0 to 24.

LEAF DENSITY

(exclude epicormics on bole)

9	very dense
7	dense
5	average (refers to ideal, not the stand)
3	sparse
1	very sparse
0.5	all leaves dead
0	leaves absent

If much fruit is present, increase the score by 1.

INCIDENCE OF DEAD BRANCHES

9	none present
8	< 50% of branchlets dead
7	≥ 50% of branchlets dead
6	< 50% of small branches dead
5	≥ 50% of small branches dead
4	< 50% of large branches dead
3	≥ 50% of large branches dead
2	< 50% of primary branches dead (epicormics present)
1	≥ 50% of primary branches dead (no epicormics)
0	all branches dead

CONTRIBUTION OF EPICORMIC BRANCHES TO FOLIAGE

6	no epicormics present (foliage concentrated at extremities of branches)	
5	on crown or bole	< 25%
4	on crown or bole	> 25-50%
3	on crown or bole	> 50%
2	on crown and bole	< 50%
1	on crown and bole	≥ 50%
0.5	on crown and bole	100%
0	tree is dead	

CHAPTER 6

The Ecological Implications of Disease in the Southern Forest of South-Western Australia

B. L. Shearer

Abstract

The ecology of diseases occurring in the forests of south-western Australia are described under five broad types (root cankers, wood rots, stem and branch cankers, leaf spots and diseases of unknown etiology). *Phytophthora cinnamomi* and *P. citricola* are the main pathogens causing root cankers. Recent research has meant that relatively more is known about the distribution and impact of *Armillaria luteobubalina* in forests of south-western Australia than other wood rotting fungi. Even though the incidence of eucalypt dieback-decline has increased in south-western Australia since the 1970s, the contribution of canker fungi to stem and branch death has largely been ignored. There has been no systematic investigation of leaf spot fungi on forest trees in south-western Australia. Brown wood of karri and dieback-decline of marri and wandoo are diseases of unknown etiology. The diseases in southern forests form a complex of interactions between host, pathogen, environment and human activity. However, information is lacking on the specific requirements for pathogen survival, sporulation and spread as well as host infection and susceptibility in the environments experienced in southern forests. Implications for management are discussed.

CURRENT KNOWLEDGE

Introduction

Studies on forest diseases have mainly emphasized the effects of disease on the establishment, growth and productivity of economically important tree species. Often ignored and little studied has been the role plant diseases play in the functioning of forest ecosystems (Edmonds and Sollins 1974). Plant diseases are more often studied from an autecological (the behaviour of individual species or populations in relation to their environment) rather than a synecological or community dynamics perspective.

The pathogens reported causing diseases of the main *Eucalyptus* species occurring in forest south of the Preston River in south-western Australia are listed in Table 1. Lack of information precludes compilation of a similar list for understorey species, although the effect of disease on the understorey will be described where appropriate later. The reported pathogenic organisms in Table 1 are all fungi. This is consistent with the experience in other areas that most diseases of forest trees are caused by fungal pathogens (Schmidt 1978). Most of the discussion will relate to disease caused by fungi; disease caused by abiotic factors will be briefly covered in the section on diseases of unknown etiology.

Fungi are important in the cycling of carbon, water and nutrients within forest ecosystems, although their role in these cycles has been little quantified (Edmonds and Sollins 1974). Fungi lack chlorophyll and are primary consumers (heterotrophs) of plants as parasites or decomposers within the characteristic trophic structure of a forest ecosystem. Some fungi take on different trophic roles during different stages of their life cycle.

Plant pathogens, like other primary consumers, utilize energy from plants by destroying seeds, buds, leaves, stems or roots. In this manner plant pathogens may selectively affect competitive vigour or kill individual plants. However, the potential deleterious affects of pathogens on the vigour of native flora is often ignored. The influence of plant pathogens on intra- and inter-specific competition, the distribution of plant species, the genetic structure of populations and the diversity of individual plant communities is poorly understood (Burdon 1987).

Table 1

Diseases of the *Eucalyptus* species occurring in the Woodchip Licence Area in southwestern Australia. X indicates occurrence of disease, susceptibility obtained from inoculation or observation indicated in parenthesis and reference number in superscript.

Disease/Pathogen	Marri	Karri	Red Flowering Gum	Tingle Yellow	Tingle Red	Jarrah	Wandoo
SOUTH-WESTERN AUSTRALIA							
ROOT CANKERS							
Phytophthora cinnamomi	$X(R)^{o,y}$	$(R)^{l,y}$		$(R)^{o}$	$(R)^{o}$	$X(MS)^{o,y}$	$(R)^{y}$
Phytophthora citricola	$(MS)^{r}$					$(MS)^{k,r}$	
Phytophthora drechsleri						X^{s}	
Phytophthora megasperma var. *sojae*						$(R\text{-}MS)^{k,r}$	
Phytophthora nicotianae						X^{c}	
Phytophthora nicotianae var. *parasitica*						$(MS)^{k,r}$	
WOOD ROTS							
Armillaria luteobubalina (White rot)	$X(MS)^{h,n,p}$	$X(MS\text{-}MR)^{w,n}$				$X(MS\text{-}MR)^{h,p}$	$X(S)^{p}$
Phellinus gilvus	$X^{g,x}$					$X^{g,x}$	
Phellinus rimosus (White pocket heart rot)							$X^{g,x}$
Piptoporus australiensis (Brown cubical heart rot)		$X^{g,x}$		$X^{g,x}$			
Piptoporus portentosus (Brown rot)	$X^{g,x}$					$X^{g,x}$	
Polyporus pelliculosus (Brown powdery rot)					$X^{g,x}$		
Polyporus tumulosus (Heart rot)		$X?^{g,x}$				$X^{g,x}$	
Poria mutans (Yellow straw rot)						$X^{g,x}$	
BRANCH AND STEM CANKERS							
Botryosphaeria ribis	$X^{c}(R)^{q}$	X^{c}				$X^{c}(MS)^{q}$	X^{c}
Endothia gyrosa	X^{c}					X^{c}	X^{c}
Ramularia pitereka (*Sporotrichum destructor*?)	$X(S)^{t,z}$		$X(S)^{t,z}$				
LEAF SPOTS							
Davisionella eucalypti						X^{w}	
Fairmaniella sp.							X^{s}
Leptomelanconium australiense			X^{v}				
Phoma sp.						X^{s}	
UNKNOWN ETIOLOGY							
Brown wood		X					
Crown decline	X						X

Table 1 (*cont'd*)

Disease/Pathogen	Marri	Karri	Red Flowering Gum	Tingle Yellow	Tingle Red	Jarrah	Wandoo
OUTSIDE SOUTH-WESTERN AUSTRALIA							
WOOD ROTS							
Armillaria luteobubalina			X[u]				
Phellinus gilvus		X[i]					
STEM CANKERS							
Cytospora australiae			X[b]				
Endothia gyrosa		X[d]					
Pestalotia disseminata		X[i]					
LEAF SPOTS							
Cercospora eucalypti			X[b]				
Mycosphaerella sp.		X[i]					
Mycotherium amygdalinum			X[m]				
NURSERIES							
DAMPING OFF							
Phytophthora cinnamomi			X[e]				
Phytophthora cryptogea					X[f]		
Phytophthora sp. (unnamed)					X[e]		
STEM CANKER							
Botrytis cinerea		X[s]					
LEAF SPOTS							
Hainesia lythri		X[j]					
Sphaerotheca alchemillae (Powdery Mildew)			X(VS)[a]				

Marri (*Eucalyptus calophylla*)
Karri (*Eucalyptus diversicolor*)
Red Flowering gum (*Eucalyptus ficifolia*)
Tingle
  Yellow (*Eucalyptus guilfoylei*)
  Red (*Eucalyptus jacksonii*)
Jarrah (*E. marginata*)
Wandoo (*E. wandoo*)

S = Susceptible, R = Resistant, M = Moderate, V = Very.

References: [a]Boesewinkel (1981); [b]Browne and Laurie (1968); [c]Davison and Tay (1983); [d]D'oliveira (1931); [e]Forsberg (1985); [f]Hardy and Sivasithamparam (1988); [g]Hilton (1982); [h]Kile *et al.* (1983); [i]Lundquist and Baxter (1985); [j]Lundquist and Foreman (1986); [k]Newhook and Podger (1972); [l]Palzer and Rockel (1973); [m]Park and Keane (1982); [n]Pearce *et al.* (1986); [o]Podger (1968); [p]Shearer and Tippett (1988); [q]Shearer *et al.* (1987b); [r]Shearer *et al.* (1988); [s]Shivas (1989); [t]Smith (1970); [u]1Smith and Kile (1981); [v]Sutton (1974); [w]Swart (1988); [x]Tamblyn (1937); [y]Tippett *et al.* (1985); [z]Walker and Bertus (1971)

More is known of environmental factors affecting disease and the ecology of pathogens occurring in south-western Australia, than on the implications of disease in an ecosystem context. For this reason the diseases occurring in the forests of south-western Australia are described under five broad types (Table 1). Table 2 summarizes the life cycles and Table 3 possible implications of representative pathogens for the four disease types of known etiology. As an aid to the assessment of potential disease problems, Table 1 also includes those pathogens reported when the *Eucalyptus* species were grown outside of south-western Australia and in nurseries.

DISEASES OF JARRAH, KARRI, MARRI AND WANDOO FORESTS OF SOUTH-WESTERN AUSTRALIA

Root Cankers

Phytophthora species are the main pathogens infecting secondary phloem of roots of *Eucalyptus* in south-western Australia (Table 1). Although mainly root pathogens, *Phytophthora* species can also invade the collar and lower stem. The *Phytophthora* species in Table 1 are introduced soil borne opportunists whose life cycles are characteristically dependent on moisture for sporulation, survival and dispersal (Dell and Malajczuk 1989; Shearer and Tippett 1989). The soils and topography in conjunction with the hydrological cycle and a susceptible plant community of south-western Australia have provided niches whereby *Phytophthora* species can survive dry conditions, despite the harsh dry summers experienced in the region. The interactions that have created the diversity of microenvironments and conditions favourable for sporulation, survival and dispersal of *Phytophthora* species in south-western Australia are detailed in Shearer and Tippett (1989) and can only briefly be described here.

Phytophthora species are probably *r* strategists in the *r-K* continuum of MacArthur and Wilson (1967) as they can quickly reproduce and complete their life cycles (Table 2) when conditions are favourable. The sporangium-zoospore cycle can rapidly produce and release large numbers of zoospores (Table 2) into the soil when conditions are warm and moist. Favourable conditions for sporulation in the mediterranean climate experienced in south-western Australia, are most likely to occur in autumn and spring. Low temperatures in winter and low moisture levels in summer inhibit sporulation. Conditions for survival are not as restrictive as those for sporulation and the fungi survive wherever the soil is moist or infected host tissue is present. Sexual production of oospores is an important form of reproduction of homothallic species such as *P. citricola* and *P. megasperma* var. *sojae* because the thick-walled spores are more resistant to desiccation than zoospores.

Favourable environments for sporulation and survival vary between areas and from year to year. The coincidence of warm moist conditions depends on the commencement and finishing of frontal winter rains, the frequency of summer rain, the occurrence of water-gaining areas from near-surface seepage above impeding horizons within the soil profile, the amount of soil cover, and of disturbance. The distribution of susceptible host material is also an important factor.

Infected moist soil moved by human activity is the main way the *Phytophthora* species are dispersed over large distances. Natural dispersal occurs by water flowing in surface and near-surface drainage systems and by growth through root systems (Table 2).

Phytophthora cinnamomi and *P. citricola* are the most frequently recovered and widely distributed *Phytophthora* species in the southern forest region (Table 4). The two *Phytophthora* species have been recovered mainly from jarrah (*E. marginata*) forest, but also from karri (*E. diversicolor*) forest (Schuster 1978; Stukely and Hill[1], personal communication; unpublished reports by CALM's dieback mapping group). Often symptom expression is subtle, resulting in underestimation of the occurrence of the *Phytophthora* species in southern forests. Disease may only be evident as sparsely scattered deaths of susceptible hosts. Large areas are classified as uninterpretable (Table 4) because of either the lack of visible susceptible indicator species on aerial photographs or resistant species have colonized and dominate infected areas in which the susceptible vegetation has been killed. Checks on the ground may improve the precision of interpretation by identifying infection in the shrub layer not evident on aerial photographs. An area infected with *P. cinnamomi* was mapped through a karri stand in this manner (Cell 13 Report by CALM's dieback mapping group). Unrecognized infections in uninterpretable areas have the potential of providing sources of inoculum for inadvertent spread.

Infections associated with *P. citricola* have been noted as part of mapping for *P. cinnamomi*, but the areas infected by *P. citricola* have not been determined. Impact of *P. citricola* is confined to isolated deaths in the shrub layer. However, little is known of the susceptibility of native flora to *P. citricola* infection.

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[1] M.J. Stukely and T. Hill, CALM, Como

Table 2

Summary of the ecology of selected fungal pathogens occurring in forests of south-western Australia

	Disease/Pathogen				
Factor	Root cankers (*Phytophthora cinnamomi*)	Wood rots (*Armillaria luteobubalina*)	(*Piptoporus australiensis*)	Stem cankers (*Botryosphaeria ribis*)	Leaf spots
Taxonomic class[a]	Oomycetes	Basidiomycetes	Basidiomycetes	Ascomycetes Deuteromycetes	Ascomycetes Deuteromycetes
Introduced	yes	no	no	yes	?
Reproduction	zoospores	basidiospores	basidiospores	conidia	conidia
Primary disease cycle	<yr	<1- many yr	<1- many yr	<1 yr	<1 yr
Infectious period	<wk-month	1-many months	1-many months	<wk-month	<wk-month
r/*K* strategy	*r*	intermediate	*K*	*r*	*r*
Dispersal	water, soil, root	air, root	air	air, rain-splash	air, rain-splash
Infection	direct/wound	wound/direct	wound	wound/direct	wound/direct
Predisposing factors	moisture defoliation(?) nutrition	drought defoliation nutrition	nutrition(?)	drought defoliation nutrition	moisture nutrition

[a] Alexopoulos (1962)
? = insufficient information

The ecology of *P. citricola* in forests of south-western Australia is poorly understood. *Phytophthora citricola* has a lower temperature optimum for growth in jarrah roots than *P. cinnamomi* (Shearer *et al.* 1987a). Low temperature in winter and autumn may inhibit growth of *P. citricola* less than that of *P. cinnamomi*. Inoculation studies have shown that jarrah is equally susceptible to invasion by *P. citricola* and *P. cinnamomi* (Shearer *et al.* 1988). In contrast, marri (*E. callophylla*) is more susceptible and *B. grandis* more resistant to invasion by *P. citricola* than by *P. cinnamomi*. Lesions of *P. citricola* in *B. grandis* are likely to be confined. Thus infection of the secondary tissue of *B. grandis* is unlikely to be as favourable for survival and dispersal of *P. citricola* as it is for *P. cinnamomi* (Shearer *et al.* 1988). Spread of *P. citricola* must be prevented because the fungus has been frequently recovered from soil in landings and roadside drains, the fungus produces oospores resistant to adverse conditions and the potential host range of the pathogen is poorly understood.

Phytophthora cinnamomi is widely distributed throughout the southern forest (Table 4). Infections are mainly associated with drainage lines or road making and logging activity prior to 1970 (CALM's dieback Mapping group Cell Reports).

Impact of *P. cinnamomi* is mainly confined to death of the shrub layer. Many of the species of the families Proteaceae, Epacridaceae, Dilleniaceae and Myrtaceae make up a large component of the jarrah forest understorey and are susceptible to infection by *P. cinnamomi.* Despite this, the changes in understorey composition caused by *P. cinnamomi* infection are poorly documented for forests of south-western Australia (Shearer and Tippett 1989). In particular, more information is needed on changes in plant species richness following infection and the associated affects on fauna.

Jarrah is the only *Eucalyptus* species in the overstorey susceptible to *P. cinnamomi* (Table 1) and death of jarrah is often confined to isolated clumps on lateritic or poorly drained soils. Most of the jarrah deaths are described as old in the CALM's dieback mapping group Cell Reports, but the time of death is not known. A report by Podger in 1966 (report on a field examination of southern dieback areas, unpublished report of the CSIRO Forest Research Institute, Kelmscott) dates one period when jarrah died in infected southern forest areas.

Table 3

Summary of the implications of infection of selected fungal pathogens occurring in forests of south-western Australia

	Disease/Pathogen				
Factor	Root cankers	Wood rots		Stem cankers	Leaf spots
	(*Phytophthora cinnamomi*)	(*Armillaria luteobubalina*)	(*Piptoporus australiensis*)	(*Botryosphaeria ribis*)	
Host range	very wide	very wide	narrow	wide	narrow?
Impact on:					
Understorey	death	death	none	?	?
Regeneration	death	death	none	?	?
Overstorey	decline & death	decline & death	decline & death	decline & death	decline
Effect on diversity	reduction	reduction	none	?	?
Part of natural succession	not in past, but now an influence	yes	yes	probably[a]	probably
Occurrence of damage in host	all stages	all stages	thinning,harvest rotation, utilization	all stages	all stages
Affects growth of *Eucalyptus* host	reduce(?)	reduce(?)	none(?)	reduce(?)	none(?)
Canopy reduction	yes	yes	no	yes	some
Influences energy & nutrient cycling	yes	yes	yes	yes	yes
Hazard rating	yes	no	no	no	no
Rating of risk of infection	possible	no	no	no	no
Control options:					
Minimize disturbance	yes	yes	yes	yes	yes
Current hygiene practices	applicable	applicable	n/a	n/a	n/a
Reduction by fire of:	susceptible hosts	inoculum	inoculum	inoculum	inoculum
Reduction by antagonists of:	inoculum	substrate & inoculum	substrate & inoculum	?	?
Enhance host resistance	?	reduce stress	reduce stress	reduce stress	reduce stress
Fertilization	?	?	?	?	?
Chemical	spot infections	n/a	n/a	n/a	n/a
Resistance	in rehabilitation	in rehabilitation	?	?	?

? = insufficient information
n/a = not applicable
[a] endemic fungi would

Research conducted in the northern jarrah forest provides most of the information on the ecology of *P. cinnamomi* in south-western Australia (Shearer and Tippett 1989). Less research has been done on *P. cinnamomi* in southern forests; this is documented in four publications (Palzer and Rockel 1973; Christensen 1975; Schuster 1978; Strelein 1988) and a report (Grant and Blankendaal 1988). Because of differences in climate, landform, soils, vegetation and the intensity of human activity between the regions, results from research in northern forest cannot be extrapolated south of the Preston River (Shearer and Tippett 1989).

Christensen (1975) monitored moisture and temperature in the top 75 mm of soil and found that removal of canopy, scrub or litter cover from the soil increased the coincidence of warm moist conditions

Table 4

Incidence of *Phytophthora cinnamomi*, *Phytophthora citricola* and *Armillaria luteobubalina* in southern jarrah forest compiled from maps and reports of the Dieback Mapping Group, Inventory and Planning

Block	Area (ha)	Year Interpreted	*Phytophthora cinnamomi*			*Phytophthora citricola*	*Armillaria luteobubalina*
			Area interpretable (%)	Interpreted area infected (%)	Infections (No.)	Infections (No.)	Infections (No.)
Carey	2195	1982	52	37	80	0	0
Coonan-Warrup	8100	1982	-	-	20	0	25
Iffley-Strickland	2535	1982	81	53	62	2	1
Easter-Andrew-Barlee	9352	1982/83	54	31	152	12	38
Mattaband-Burnside	5520	1983	62	44	15	2	2
Spring-Poorginup	3910	1984	35	72	22	0	6
Carter-Thornton-Yornup-Alco	9750	1984/85	-	-	56	15	16
Crowea	80	1986	-	-	9	0	0
Dordagup	308	1986	66	8	55	0	2
Mack	296	1986	-	-	3	0	0
Wye-Deep-Sharpe	15250	-	-	-	118	0	6

- no data

favourable for disease development. A small increase in temperature towards the end of winter increased the duration of the period when conditions were favourable for development of *P. cinnamomi* in spring. However, the results of Christensen's (1975) study underestimate the likely effects of environment on development of *P. cinnamomi* in the southern region and differences between regions. Temperature and moisture were monitored at only four sites over 18 months during a period of below-average rainfall. Furthermore, the existence of an environment favourable to sporulation, survival and dispersal of *P. cinnamomi* at depth below the soil surface was not appreciated at the time and the study did not include the potential effects of summer rain on disease development. No determination has been made of the population dynamics of *P. cinnamomi* in the range of environments and soils of southern forest. Information is lacking on the specific requirements for pathogen survival, sporulation and spread as well as host infection and susceptibility in southern forest soils. This information is needed to predict the development of *P. cinnamomi* in the range of environments experienced in southern forest and the effects of disturbance.

Southern forest sites that are likely to be favourable for increase and spread of *P. cinnamomi* and host infection can be identified from the work of Schuster (1978), Strelein (1988) and Grant and Blankendaal (1988). Schuster (1978) observed greatest impact of *P. cinnamomi* in the overstorey in sites with shallow duplex and gravelly-sand lateritic soil. In the development of a site-vegetation classification system for the southern jarrah forest, Strelein (1988) found greatest incidence of *P. cinnamomi* in less fertile, high rainfall, poorly drained sites associated with southern jarrah site-vegetation types P, R and S.

Grant and Blankendaal (1988) have used Strelein's site-vegetation types and landform to develop a hazard rating system for *P. cinnamomi* in southern jarrah forest. High hazard was associated with southern types P and S and landform units Dwellingup, Trent, Ellis, Mungardup and Caldyanup. These upland types or landform units are associated with lateritic or duplex soils that impede the drainage of water. Low hazard was associated with types T, S and K and the Crowea and Bevan landform units in dissected valleys with fertile loamy soils. Both high and low hazard were associated with the following landform units: Hester on ridge crests with gravelly-sand over duricrust; the units Collis, Mattaband and Keystone with yellow duplex soils; and the stream unit in minor valleys with swampy floors. The hazard rating system has only recently been introduced and mapping commenced. Mapping of hazard must have high priority to accurately determine the proportion of southern forest at risk from *P. cinnamomi* infection.

It is generally considered that impact of *P. cinnamomi* is less in southern jarrah forest than in northern jarrah forest (Strelein 1988). Reasons for this difference are based mainly on empirical observation. It cannot be implied from the difference between the two regions that the implications of *P. cinnamomi* infection is less in southern than northern forest. Interactions between historical and environmental factors have effected differences in disease incidence and impact between the two regions.

Historically, southern jarrah forest has been exposed to less disturbance that favours spread and intensification of *P. cinnamomi* than northern forest. Most of the northern forest has been logged at least two or three times (Abbott and Loneragan 1986) while southern forest has mainly been logged only once. Southern forest has not been exposed to the widespread mineral exploration and the road construction and earth movement associated with bauxite mining carried out in northern forest.

Strelein (1988) and Christensen (1975) suggest that disturbance from more intensive management may lead to intensification of disease owing to *P. cinnamomi* in southern forest. Accurate predictions are difficult, however, without more information on the influence of the microclimates experienced in southern forest on the development of *P. cinnamomi*. Furthermore, the use of integrated control can minimize the effects of disturbance on *P. cinnamomi* intensification and spread. Hygiene procedures already in practice (Underwood and Murch 1984) need to be integrated with control methods that reduce the rate of disease development. However, integrated control measures require further development before practical application (Shearer and Tippett 1989).

Localized death of jarrah has occurred in infected areas of southern forest, but little is known of the timing of these deaths and the climatic conditions when they occurred. The hazard rating system of Grant and Blankendaal (1988) allows the mapping of high hazard sites, but cannot predict how quickly disease will be expressed following infection. Prediction of the rate of disease expression in sites of different hazard ratings needs a better understanding, than is presently available, of the occurrence of conditions favourable for *P. cinnamomi* sporulation and survival and host infection in southern forest sites, especially at depth in the soil profile.

Host studies have shown that site factors affect the susceptibility of jarrah to invasion by *P. cinnamomi* (Tippett *et al.* 1987). The susceptibility of jarrah on sites of different hazard ratings has yet to be assessed. Insufficient is known of the physiological status of host plants on different sites under different climatic and management conditions to determine when and for how long some species are vulnerable to infection and invasion by *P. cinnamomi* (Shearer and Tippett 1989). On intermediate hazard sites where jarrah is likely to survive, the effect of *P. cinnamomi* infection on regeneration, growth and leaf area is poorly understood (Table 3).

Wood Rots

Wood rotting fungi have cellulolytic and lignolitic enzymes and are important contributors to nutrient cycling within an ecosystem through the decomposition of plant material including standing timber. Table 1 lists only decay organisms that attack the sapwood or heartwood of standing trees. The volume of timber lost through decay has not been determined for forests of south-western Australia, despite the growing appreciation of the need for better utilization and conservation of the present timber resource.

Little research has been undertaken on the ecology of wood rots caused by basidiomycetes (Table 1) that are native to south-western Australia (Table 2). Tamblyn (1937) determined the identity and incidence of a number of decay fungi in south-western forests (Table 1) and the taxonomy was updated by Hilton (1982, 1988). A general description of wood rotting fungi attacking standing jarrah is given by Hilton *et al.* (1989).

Although specific details are lacking, the life cycles of the wood rotting fungi occurring in forests of south-western Australia (Table 2) are comparable to similar organisms in other forests. Wood rotting fungi can be considered as intermediate or *K* strategists in the *r-K* continuum (MacArther and Wilson 1967) as their life cycles are mainly dependent on the ability to maintain stable population carrying capacity (*K*) from annual reproduction (Table 2). This is in contrast to the *r* strategists, such as the causal fungi of root and stem cankers, that have fluctuating population levels and rapid reproduction (Table 2) in response to transient favourable conditions in a fluctuating environment.

Infection from wood decay fungi occurs mainly from aerial dispersed basidiospores or through mycelium in root systems. Basidiospores, formed by sexual recombination of gametes, are shed in winter from annual or perennial fruiting bodies found on decayed roots and stems of dead and living trees. Fruiting bodies of *A. luteobubalina* are mainly produced in June and July (Pearce *et al.* 1986; Shearer and Tippett 1988), but the environmental stimuli favouring fruiting at this time of the year are poorly understood. Dead trees following wildfire, or karri stumps left after logging may provide a long-term food base for sporulation of *Phelinus* species that cause serious heartwood decay of standing trees (Pearce and Malajczuk 1990a).

Infection points for entry of germinating basidiospores are mainly provided by wounds caused by fire, logging injuries, dead or broken limbs or insect damage. The effect of environment on the survival and infectivity of basidiospores of most decay fungi is poorly understood (Merrill 1970) and has not been determined in south-western Australian forests. The distribution of infection points and aerial dispersed basidiospores results in a discontinuous, discrete distribution of infections of different genotypes. The number and distribution of different genotypes for a particular decay organism can provide an estimate of the frequency of infection from basidiospores, but no analysis of this type has been done in forests of south-western Australia.

Recent research on *Armillaria luteobubalina* in the karri and jarrah forest (Kile *et al.* 1983; Pearce *et al.* 1986; Shearer and Tippett 1988) has meant that relatively more is known about the distribution and impact of *A. luteobubalina* in forests of south-western Australia than the other wood rotting fungi. *Armillaria luteobubalina* is a primary pathogen widely distributed throughout south-western Australia (Pearce *et al.* 1986; Shearer and Tippett 1988; Table 4). The location, but not the size of infection centres have been determined during routine mapping (Table 4) and the area of forest affected by *A. luteobubalina* is not known. The pathogen spreads within infection centres by mycelial growth through roots. New infections are established by contact between roots and stems, and dead roots and stumps increase the inoculum level. Disease caused by *A. luteobubalina* in forests in Victoria spreads at a rate of 0.5-2.0 m/year (Kile 1981), but no estimates are available for south-western Australia.

Armillaria luteobubalina infects a wide range of plant species from diverse families (Pearce *et al.* 1986; Shearer and Tippett 1988), but changes in species richness within affected areas have not been determined. Many hosts that resist infection by *P. cinnamomi*, are susceptible to *A. luteobubalina* (Shearer and Tippett 1988).

Impact of *A. luteobubalina* in the jarrah forest varies between plant community and climatic zone (Shearer and Tippett 1988). Patch death of *E. wandoo*, *B. grandis* and *Xanthorrhoea preisii*, and crown decline and scattered death of marri and jarrah is associated with *A. luteobubalina* in intermediate-low rainfall zones receiving <1000 mm p.a. Damage to trees in the high rainfall zone (>1100 mm p.a.) of the jarrah forest is not as severe as in eastern low rainfall zone forest, despite the fact that the fungus is well established and fruits prolifically. Within high rainfall-zone infection centres, marri, jarrah and *B. grandis* exhibit crown decline but there are few and scattered mortalities. Scattered deaths of overstorey species partly reflects variation in the response to infection within a species; some individuals resist infection while others of the same species die when infection reaches the base of the stem. The reasons for this variation is not known and the contribution of stress, inoculum potential, site, soil and climatic factors or the genotype of host or pathogen to variation in response to infection has yet to be determined (Shearer and Tippett 1988).

Impact of *A. luteobubalina* in the karri forest changes with age of the forest (Pearce *et al.* 1986). The pathogen kills vigorous karri saplings in the youngest stands and a significant loss of regeneration trees can occur in infected areas. Loss in older infected stands (5-15 years old) is mainly confined to death of subdominant and suppressed trees and *A. luteobubalina* would play a role in the thinning of natural stands. The deaths occur mainly within a 10-m radius of infected karri and marri stumps.

The impact of *A. luteobubalina* in karri stands probably reflects changes in the balance between inoculum potential of the pathogen, host resistance as a function of vigour and tree size, and environmental stress during the life of the stand (Pearce *et al.* 1986). *Armillaria luteobubalina* invades the root system of stumps following logging (Kile 1981; Shearer and Tippett 1988) and the inoculum potential of the fungus is probably greatest in young generation stands. Infected stump roots represent a large source of inoculum compared with the relatively small diameter of young karri sapling roots and the inoculum potential of the fungus probably overrides host resistance factors (Pearce *et al.* 1986). Dominant trees in older stands are probably sufficiently vigorous to contain lesions of *A. luteobubalina* and prevent girdling of the root crown and death. However, infection of dominant trees, although contained, weakens roots and may predispose the trees to windfall (Pearce *et al.* 1986).

Logging prescriptions could increase the incidence of disease caused by *A. luteobubalina* in the forest by increasing the amount of inoculum present (Pearce *et al.* 1986; Kellas *et al.* 1987). The frequency of cutting mixed eucalypt forests of Victoria had a greater effect on disease development by *A. luteobubalina* than cutting intensity *per se*. Regular creation of stumps through frequent cutting increases both the inoculum level and the probability of remaining trees being in close proximity to inoculum (Kellas *et al.* 1987). However, the implications of cutting frequency also depend on the distribution of the fungus and the effect of environmental factors on disease expression.

As native organisms it is likely that the decay fungi have always caused some disease as part of the functioning of the ecosystems in which they occur (Table 3). For example, high intensity fire kills karri trees and stimulates regeneration, providing similar opportunities for *A. luteobubalina* as in cut-over forests. Nevertheless care must be taken that forest operations do not shift the balance in favour of the fungi by providing infection sites through wounding and a large food base for reproduction. Information on the distribution, incidence and ecology of fungi causing decay is needed to predict the effects of forest operations on decay organisms under different climatic and site conditions.

Stem and Branch Cankers

Even though the incidence of eucalypt dieback-decline has increased in south-western Australia since the 1970s (Kimber 1980), the contribution of canker fungi to stem and branch death has largely been ignored. Mortality and decline of marri and red flowering gum (*E. ficifolia*) was associated with stem cankers in the mid 1930s (Smith 1970). Davison and Tay (1983) identified a number of pathogenic fungi associated with stem and branch cankers of forest trees in south-western Australia. Canker fungi were associated with a complex of factors causing crown decline of wandoo (*E. wandoo*) (Albone 1989). Severe cankering of marri in southern forest is causing concern (C. Muller[2] *personal communication*), but has not been investigated.

Table 1 lists the primary pathogens that cause perennial cankers. Not included are fungi that are frequently isolated from cankers but are nonagressive facultative parasite (e.g. *Cytospora eucalypticola*). *Botryosphaeria ribis* is an aggressive pathogen widely distributed in tropical and temperate regions and able to infect at least 34 genera and 20 families of plants (Smith 1934). Whereas *B. ribis* is possibly an introduced pathogen (Davison and Tay 1983), *Ramularia pitereka* is probably native to south-western Australia (Walker and Bertus 1971). Although Davison and Tay (1983) consider *Endothia gyrosa* to be native to the region, Walker (1987) questions the native status of this pathogen.

How the canker causing fungi complete their life cycles in south-western forest is poorly understood. This is complicated by uncertainties in the identity of canker fungi. The identity of the *Endothia gyrosa* in south-western forests was recently confirmed by isozyme analysis against voucher specimens (E. Davison[3], *personal communication*). The fungus causing the canker on red flowering gum was called *Sporotrichum destructor*, but the name was never validly published and no description of the fungus exists in the literature. The fungus causing cankering of red flowering gum and marri may be *R. pitereka*, but the identity remains uncertain until fresh isolates can be examined (Walker and Bertus 1971).

Canker fungi are opportunistic *r* strategists (Table 2). They sporulate in dead bark and are dispersed as sexually produced ascospores in wind currents or asexually produced pycnidiospores in rain splash. Entry of germinating spores is probably gained through lenticels or wounds from branch stubs, broken branches and insect damage. Phloem and sapwood invasion results in sunken cracked areas on the stem that may expose the xylem and exude kino. Girdling by cankers can result in gradual decline from death of twigs and lateral branches to rapid death of leaders. No determination has been made of the losses of leaf area and tree function from stem and branch death in forests of south-western Australia. Progressive canker development may lead to death of diseased trees (Smith 1970; Shearer *et al.* 1987b).

Disease caused by canker fungi can be aggravated by transient stress factors (Schoeneweiss 1975). Trees planted outside the normal geographic range may experience environmental stress with an associated decline in resistance to infection by canker organisms (Shearer *et al.* 1987b). Stress from defoliation by insects (Abbott, this volume) can predispose trees to canker organisms (Schoeneweiss 1975).

Leaf Spots

There has been no systematic investigation of leaf spot fungi on forest trees in south-western Australia similar to studies in eastern Australia (e.g. Park and Keane 1982). Table 1 list leaf spot fungi collected and identified during the investigation of other pathogenic organisms. More species, than in Table 1, would be described from a systematic survey of leaf spots on eucalypts in south-western Australia.

In general leaf spots are opportunistic *r* strategists (Table 2). They sporulate in lesions and are dispersed as sexually produced ascospores in wind currents or asexually produced conidia in air and pycnidiospores in rain splash. Many leaf spot fungi are weak facultative pathogens and are associated with insect damage to leaves.

Most of the leaf spot fungi of eucalypts would be native to south-western Australia and play an important role in litter decomposition and nutrient cycling. Defoliation of eucalypt seedlings by host-specific leaf pathogens affect the composition of regeneration in forests of eastern Australia (Burdon and Chilvers 1974). Severe defoliation of eucalypts by leaf spots has not been reported in Western Australia. Karri seedlings are very susceptible to leaf infection by

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[2] C. Muller, CALM, Bunbury

[3] E. Davison, CALM, Como

powdery mildews in the nursery (Table 1). The effect of powdery mildew infection on karri seedling susceptibility and establishment in the field is not known.

Unknown Etiology

Brown wood of karri and dieback-decline of marri and wandoo are diseases of unknown etiology. Browning of wood occurs frequently in regrowth karri sawlogs and also occurs in logs from regrowth forests. It causes concern owing to reduced recovery and increased milling costs. The cause of brown rot and its effect on timber durability and permeability are currently being investigated.

Some areas of marri and wandoo in forests and rural areas have suffered from decline and some death over the last decade (Kimber 1980; Tippett and Shea 1985). However, investigation of the causes of dieback-decline has received little attention. Damage from frost, drought, salinity, disturbance, pests and pathogens are suggested causes, but the etiology is still uncertain. The succession of factors, such as climatic, edaphic or biotic stress that predispose trees to infection by pathogens, often complicates the determination of the cause. Deforestation through dieback-decline has obvious implications for wildlife habitats, conservation of roadside verges, amenity values such as the provision of shade and the control of salinity and erosion by reaforestation.

CONCLUSIONS

Systematic surveys are required to improve information on the identity and incidence of pathogens causing disease in forests of south-western Australia. Such surveys would identify previously unrecognized pathogens, as illustrated by the survey on cankers by Davison and Tay (1983). Ranking of the relative importance of disease would then be possible according to deleterious affects on conservation and production values.

As part of the functioning of ecosystems, pathogens continually modify the plant populations in which they occur. The effects of disease on plant populations are thus more often a reflection of persistent long-term changes than of devastating damage. Disease causing devastating damage is the exception, being more the symptom of imbalance mainly brought about by human intervention. The initiation and rate of increase and spread of disease is considered to be mitigated by the functional diversity of forest ecosystems (Schmidt 1978; Dinoor and Eshed 1984). Functional diversity comprises the characteristic structural, macro and microenvironmental and genetic diversity of a forest ecosystem that act collectively to impede disease development in space and time (Schmidt 1978). The ability of a system to cope with disease is also related to the length of time host and pathogen have coevolved. Coevolution may have a greater stabilizing role than diversity, in mitigating the effects of disease in some instances (May 1976).

Functional diversity and coevolution of host and pathogen cannot buffer ecosystems against introduced pathogens with a wide host range. At least three of the pathogens listed in Table 1 have been introduced into south-western Australia, namely *B. ribis* (Davison and Tay 1983), *E. gyrosa* (Walker 1987) and *P. cinnamomi* (Newhook and Podger 1972). *Phytophthora* species other than *P. cinnamomi* may also have been introduced into south-western Australia (Shearer and Tippett 1989). These pathogens are widely distributed in temperate and sub-tropical regions of the world and infect a wide range of plant species from diverse families. The impact of *P. cinnamomi* infection on the jarrah forest is an exceptional example of an introduced pathogen with a wide host range causing great damage to a diverse but mainly susceptible plant community (Shearer and Tippett 1989).

Predictions of the implication of disease requires an understanding of the varied life cycles of the pathogens causing disease in southern forest (Table 2). Many of the pathogens listed in Table 1 are natives of south-western Australia and play a part in the functioning of forest ecosystems (Table 3). However, the life cycles of *Phytophthora* species other than *P. cinnamomi*, and the fungi causing wood decay, stem and branch cankers, and leaf spots have not been investigated and are poorly understood in south-western Australia. The cause of a number of diseases has still to be determined (Table 1).

The diseases in southern forests form a complex of interactions between host, pathogen, environment and human activity. More information is needed on the specific requirements for pathogen survival, sporulation and spread as well as host infection and susceptibility in the environments experienced in southern forests. The effects of potential changes in climate on disease requires a much better understanding than is available at present of the life cycles of pathogens in southern forest. The effects of insect pests on life cycles of pathogens needs to be determined. Insect attack increases the incidence of wounds and stress from defoliation and predisposes trees to infection.

Human actions affect different parts of the life cycle of a pathogen to either aggravate or control disease in forest ecosystems. Human activity introduced *Phytophthora* species and *B. ribis* into diverse but susceptible plant communities of forests of south-western Australia. Movement of infected soils by human activity is an important mechanism of dispersal of *Phytophthora* species and has necessitated the commitment of considerable resources into hygiene measures to prevent and minimize this type of spread. Fungi causing wood decay, cankers and leaf spots are mainly dispersed by natural means (Table 2), but care must be taken to ensure that human activity does not cause wounds favourable to infection (Table 2). Cutting frequency and regeneration from coppice stumps can exacerbate the incidence of wood decay. Fire may cause wounds favourable for infection by decay fungi, but destroy inoculum of canker and leaf spot fungi. Stand management must aim to prevent stress and physiological imbalance that may predispose a tree species to infection.

The life cycles of different pathogens can be linked by the stimuli and interactions occurring within forest ecosystems. Care must therefore be taken to ensure that the management of a forest does not consider a few pathogens to the exclusion of others that may occur.

CURRENT RESEARCH

Identification of the organisms causing brown wood in karri is the only current research on disease being undertaken in southern forest. *Armillaria luteobubalina* is an intermediate-*K* strategist and probably sensitive to control by reducing the food base for reproduction. The use of decay fungi to reduce stumps as a food base for *A. luteobubalina* in southern forest has been investigated (Pearce and Malajczuk 1990b).

Research on diseases in the forest is mainly being carried out north of the Preston River or in woodlands and shrublands of the coastal plain. Research is continuing on the factors affecting the development and control of *P. cinnamomi* in the northern jarrah forest (Shearer and Tippett 1989) and coastal plain (Shearer and Hill 1989). Current research on *A. luteobubalina* has determined the impact of the pathogen in the wandoo forest and coastal shrublands. Methods for pathogenicity tests are being developed. A survey of dieback-decline of wandoo has been completed but the data requires analysis (Albone 1989).

There is no research in progress on the cause of dieback-decline diseases and cankering of marri. No determination is being made of the environmental factors affecting the life cycles of *Phytophthora* species, *A. luteobubalina* and other decay organisms, stem cankers and dieback-decline of unknown etiology in southern forest.

HIGH PRIORITY ADDITIONAL RESEARCH REQUIREMENTS

Shearer and Tippett (1989) detail research priority areas and questions for *P. cinnamomi* and other *Phytophthora* species. Research on diseases in southern forests needing immediate attention are listed in order of priority:

- Determine the cause of cankering and dieback decline of marri and options for control.
- Develop a data base of the susceptibility of plant species to *Phytophthora* species and determine the long-term effects of infection on community diversity.
- Determine the relationships between climatic events and the processes affecting the development of *P. cinnamomi* and *P. citricola* and host infection in soil profiles of southern forest. Use the information to apply risk and hazard rating systems.
- Identify and rank diseases in southern forest from an assessment of their impact and potential threat to forest health.

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CHAPTER 7

Aquatic Fauna of the Karri Forest

S.A. Halse and J.D. Blyth

Abstract

This paper reviews current knowledge of the aquatic fauna of the karri forest, lists existing research projects and outlines the most important additional research projects that are required. Streams in the karri forest tend to have lower gradients and slower flow rates than most streams elsewhere. The majority of the energy driving karri stream ecosystems appears to come from the breakdown of vegetation growing outside the stream. Nine species of fish and several hundred species of aquatic invertebrate occur in karri streams. The invertebrate fauna is poorly known. Studies elsewhere have shown that land disturbance can affect the biology of streams and recent research has shown that, unless appropriate prescriptions are used, timber harvesting can affect the community composition of streams in the karri forest. Research topics that should receive high priority in the karri forest are an inventory of aquatic invertebrates, more detailed examination of the impact of timber harvesting and regeneration, identification of the sources of energy that drive stream processes, measurement of bed structure and sediment loads in streams, and documentation of the life cycles of fish.

INTRODUCTION

There is little information available about the aquatic fauna of the karri forest (see Christensen 1986). As a result, this review makes extensive use of relevant studies elsewhere in Australia and overseas in an attempt to determine any important biological features of the karri forest stream system that may be influenced by forest management. The major topics that we cover are:

(1) the types of streams that occur in the karri forest and their physical attributes;

(2) sources of energy input into the streams;

(3) the distribution and ecology of the animals inhabiting the streams, including their conservation status;

(4) the effects of forest management on the streams and their fauna.

PAST RESEARCH AND CURRENT KNOWLEDGE

Stream Types

Stream headwaters usually consist of a seepage area where groundwater is coming to the surface. As water begins to follow a distinct drainage line it constitutes a first-order stream. First-order streams have no tributaries, second-order streams are formed by the confluence of two first-order streams, third-order streams by the confluence of two second-order ones and so on. Rivers are usually at least fifth-order (Hynes 1970).

A recent survey of the karri forest by the authors showed that the headwaters of streams therein are mostly low-gradient swampy areas with soils that are rich in organic matter and covered with leaf litter. The main forest trees, *Eucalyptus diversicolor* and *E. calophylla*, rarely occur in headwaters; instead there is a closed canopy of smaller tree and shrub species, including *Melaleuca* spp, and sedges and rushes occur in the understorey. Free water is found on the surface only in winter and spring although the water table is close to the surface throughout the year.

First-order streams dry out over summer unless they are particularly long and in a high rainfall zone; however, groundwater occurs close under the stream bed. Most first-order streams are low-gradient and swampy. They usually have a clay base in their upper reaches, near the stream headwaters, but the substratum becomes sandy a few hundred metres downstream. Flow rates are usually moderate to slow. In some cases *E. diversicolor* and other eucalypts grow to the edge of the stream; elsewhere the headwater vegetation extends along the stream. It is common for rushes and sedges to grow in the stream bed. Sometimes submerged macrophytes occur, although the extent of their occurrence is probably determined largely by the openness of the canopy and the degree of organic staining in the water.

Second-order streams are similar to first-order ones except that some of them are permanent, albeit extremely slow-flowing in summer (most dry out). During winter and spring they flow faster and frequently reach 0.5 m in depth. As a result they often have a more pronounced stream channel than first-order streams.

Third-order and fourth-order streams are permanent, reach depths of up to 1 m but otherwise are similar to second-order streams.

Rivers are essentially similar to streams except that they are larger. They have sandy substrata and often have steep, eroded banks covered in sedges, rushes and shrubs. Water flow is usually moderate and they can be several metres deep. *Agonis flexuosa* and *A. junerperina* trees usually grow along the edge of the river on a flat 'terrace' that occurs either side before the land rises to the surrounding forest.

The above description covers the 'typical' karri forest stream types. They are not typical of forest streams elsewhere in Australia, or the rest of the world, where gradients are often much greater, which leads to higher water velocities that cause erosion of the stream bed with fine sediment and sand being scoured away to expose pebbles and boulders (Hynes 1970; Bayly and Williams 1973). Sections of eroded 'riffle-zone' streams do occur in the karri (e.g. in Carey Brook,); they are generally second and third-order streams, although waterfalls (e.g. Beedelup National Park) provide extreme examples of eroded areas on higher-order streams.

In the southern and south-western part of its distribution karri grows in an even flatter landscape. The streams in these areas often flow through extensive areas of heath that occur on low sandy 'flats', while karri grows on the gentle hills. These streams usually have very darkly stained water and there is a shrubbier and taxonomically-different riparian vegetation.

The unusual nature of karri forest streams means that it is often inappropriate to extrapolate results from eastern Australian, and overseas, studies to Western Australia.

Energy Sources

The energy that drives ecological processes in streams is derived from two major sources: direct sunlight that stimulates the growth of photosynthetic aquatic plants (algae and macrophytes) in the stream; or allochthonous organic material that has been produced outside the stream and is dropped or washed into it. Allochthonous material consists of terrestrial plant matter in various stages of breakdown. Streams deriving most of their energy from photosynthesis within the stream (i.e. photosynthesis, P, is greater than respiration, R) are called autotrophic while those in which most of the energy comes from allochthonous material ($R > P$) are called heterotrophic.

Sources of energy in karri forest streams have not been studied. Instream primary production has been found to be unimportant in northern hemisphere forested streams. This has also been found to be the case in eastern Australia (Blackburn and Petr 1979). It is likely that allochthonous production is the major energy source in the low-order streams that occur in the karri forest; the larger streams and rivers draining them may rely on photosynthetic production to a greater extent because there is a general trend for the relative importance of instream production to increase as the stream channel becomes more open to sunlight (Vannote *et al.* 1980).

In most studies of the input of allochthonous material into streams three size classes of material are recognized (Bunn 1986a):

(1) coarse particulate organic matter (CPOM ≥ 1 mm);

(2) fine particulate organic matter ($0.5\ \mu m <$ FPOM < 1 mm);

(3) dissolved organic matter (DOM $< 0.5\ \mu m$).

CPOM comprises mostly leaf litter, fruits, twigs, bark and branches. In Australia most CPOM enters streams in summer and is of comparatively poor quality (Bunn 1986a). *Eucalyptus marginata* is the only species that has been studied in Western Australia in terms of CPOM input into streams and its leaves have extremely long processing times and low nutritional content (Bunn 1986a). *Eucalyptus diversicolor* leaves are probably less refractile than those of *E. marginata* but still of poor quality by world standards (D.H.D. Edward[1] personal communication).

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[1] Dr D.H.D. Edward, Department of Zoology, The University of Western Australia.

There has been little study of sources of DOM and FPOM input into streams in Australia but it has been shown that they can comprise a substantial component of the total allochthonous input in northern hemisphere streams; Fisher and Likens (1973) found that 25 per cent of allochthonous input into a forested stream entered as DOM in the groundwater and 31 per cent entered as FPOM and DOM from the stream headwaters. Input of FPOM increases during times, and in situations, of high runoff. In agricultural areas, especially after heavy summer rain, this can lead to eutrophication and anoxia in rivers (Bunn 1986a). The same phenomenon has been reported in farm dams in Western Australia (Morrissy 1974).

There are no data available about the relative importance of FPOM and DOM input in Western Australian streams although D.H.D. Edward (personal communication) suggests that because of the refractile nature of eucalypt leaves, the small size of most invertebrate species and the lack of shredders in the jarrah forest, the input of FPOM and DOM from stream headwaters is the most important source of energy in jarrah forest streams. This organic matter is derived principally from the breakdown of sedges and other litter in the seasonally-wet headwaters.

Work by I.O. Growns[2] (unpublished data) has shown that the amphipod *Perthia* cf *acutitelson*, which grows up to 10 mm long, occurs in some karri forest streams at densities up to 600 m^{-2}. Large numbers of trichopterans and tipulids also occur. All these animals feed on CPOM and their presence suggests that CPOM is a more important energy source in the karri than jarrah forest and may be at least as significant as FPOM and DOM.

Aquatic Fauna

Faunal composition and conservation status

Although in general terms the same groups of animals occur in streams throughout the world, relative proportions of the groups and the actual species vary across continents and within regions. The limited work that has been done suggests the faunas of jarrah and karri forest streams are similar (Tables 1 and 2; I.O. Growns and D.H.D. Edward personal communication) although karri streams appear to contain more species. Edward believes more sampling will show that the karri forest acts as a refugium for aquatic macroinvertebrates and, hence, contains the jarrah stream fauna plus additional species restricted to karri streams, including many Gondwanaland relicts. Limited collecting by I.O. Growns, S.E. Bunn[3] and B. Knott[4] (personal communication) supports this view: all have found additional species, usually undescribed, that appear to be restricted to karri forest streams. *Kosrheithrus boorarus* and *Plectrotarsus minor* and an undescribed ephemeropteran from Carey Brook are examples.

Approximately 260 species of aquatic invertebrate have been collected in jarrah forest streams (D.H.D. Edward personal communication). An idea of the community composition may be gleaned from the Appendix in Bunn *et al.* (1986) which listed 145 species (Table 3). The fauna was dominated by insects, especially chironomids and trichopterans. Coleopterans, ephemeropterans and simuliids were the next most common groups. However, faunal composition varies markedly between the stream headwaters of a river and its larger channels (see Vannote *et al.* 1980), and Table 3 reflects only faunal composition of low-order streams.

Table 3 also gives a preliminary idea of the composition of the invertebrate fauna of karri forest streams, although one of the karri communities sampled contained a strong pond element. Like the jarrah forest streams, those in the karri are dominated by insects with chironomids and trichopterans being the most common groups. Coleoptera and tipulids appear to be the next most common groups. Even taking into account that some ostracods collected in the karri forest came from ponds, karri forest streams appear to contain more crustacea than those in the jarrah forest (although this may reflect that more sampling was done in stream headwaters in the karri).

Because the samples on which Table 3 is based were collected mainly from lower-order streams it omits decapod crustaceans which constitute the most important animal group, in terms of biomass, in most rivers and larger streams. Five species occur in the karri forest area: the marron (*Cherax tenuimanus*) in rivers; gilgies (*Cherax quinquecarinatus* and *Cherax crassimanus*) mostly in streams, which may be temporary; and koonacs (*Cherax plejebus* and *Cherax glaber*) mostly in swamps on the flats to the south of the karri forest although they can also occur on the banks of streams. [The taxonomy of *Cherax* species is controversial. Riek's (1967) classification was recently revised by Austin (1979, 1986) in unpublished Honours and PhD theses.]

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Table 1

Macroinvertebrates recorded from karri forest streams and pools connected to those streams on the southern acidic flats (Pusey and Edward 1990a). Taxa that have been recorded in jarrah forest streams are marked *, taxa not recorded in jarrah streams but widespread in the south west are marked $^{+}$. N.B. Some of the species (e.g. probably all the Hydracarina, all the Cladocera, most Collembola and some Ostracoda) are still-water species and, hence, unlikely to occur in karri or jarrah forest streams.

Taxonomic group	Common name	Species
Mollusca	Snails and mussels	*Glacidorbis occidentalis**
		*Ferrissia pettardi**
		Physastra sp*
Annelida	Worms and leeches	Oligochaeta sp
Hydracarina	Water mites	Oribatida sp 1
		Oribatida sp 2
		Oribatida sp 3
		Bdellidae sp
		Mesostigmata sp
		undescribed genus
		Limnesia sp
		Oxus sp
		Arrenurus sp
		Coaustraliobates sp
		Koenikea sp
Ostracoda	-	*Ilyodromus* sp 1*
		Ilyodromus sp 2*
		Ilyodromus sp 3
		Ilyodromus sp 4
		Gomphodella sp
		Cypretta sp
		Newnhamia sp*
		Limnocythere mowbrayensis^{+}
		Limnocythere sp
		undescribed genus
Copepoda	-	Harpacticoidea sp 1
		Calamoecia attenuata^{+}
		*C. tasmanica subattenuata**
		*Macrocyclops albidus**
		*Metacyclops cf arnaudi**
Cladocera	Water fleas	*Biapertura affinis*$^{+}$
		Echinisca cf *capensis*
		Neothrix armata^{+}

Table 1 (*continued*)

Taxonomic group	Common name	Species
		Simocephalus acutirostratus^{+} *Chydorus* sp
Isopoda	-	*Amphisopus ?lintoni* *Amphisopus ?annectans*
Amphipoda	-	*Perthia acutitelson** *Perthia* sp 1 *Perthia* sp 2 undescribed genus
Decapoda	Crayfish	*Cherax ?plebejus**
Collembola	Springtails	*Sminthurides ? stagnalis* *Xenylla* sp *Brachystomella* sp *Isotoma* sp
Zygoptera	Damselflies	Lestidae sp*
Ephemeroptera	Mayflies	*Bibulena kadjina** *Nyungara bunni**
Plecoptera	Stoneflies	*Newmanoperla exigua** *Leptoperla australica** Gripopterygidae sp
Trichoptera	Caddisflies	*Lectrides parilis** *Triplectides* sp B* *Oecetis* sp* *Notalina* sp* *Notalina* sp A* *Symphitoneuria* sp 1 *Symphitoneuria* sp 2 Leptoceridae sp 1 *Acroptila ?globosa** *Oxyethira* sp* *Hellyethira/Acritoptila* sp* *Hellyethira* sp B *Maydenoptila* sp A* *Maydenoptila* sp* *Ecnomina ?trulla** *Ecnomus pansus**

Table 1 (*continued*)

Taxonomic group	Common name	Species
		Ecnomidae sp* *Kosrheithrus* sp
Hemiptera	Bugs	*Diaprepocoris* sp *Notonecta* sp
Coleoptera	Beetles	*Sternopriscus browni** *S. marginatus** *Sternopriscus* sp 1 *Homeodytes scutellaris** *Rhantus suturalis** *Uvarus pictus** *Antiporus* sp* *Helodidae* sp* Hydrophilidae sp 1* Hydrophilidae sp 2 Curculionidae sp 1 Curculionidae sp 2
Simuliidae	Blackflies	*Austrosimulium furiosum** *Austrosimulium* sp A* ?*Cnephia* sp 1* ?*Cnephia* sp 2*
Ceratopogonidae	-	Ceratopogonidae sp A* Ceratopogonidae sp F* Ceratopogonidae sp G*
Tipulidae	-	*Limnophila* sp
Muscidae	Flies	*Limnophora* sp*
Culicidae	Mosquitoes	*Anopheles* sp* *Aedes* sp
Chironomidae	Midges	*Aphroteniella filicornis** *Aphroteniella* sp* *Chironomus* aff *alternans** *Cladopelma curtivalva** *Cladopelma* sp *Cryptochironomus griseidorsum** *Dicrotendipes conjunctus* *Dicrotendipes* sp*

Table 1 (*continued*)

Taxonomic group	Common name	Species
		Harnischia sp*
		*Kiefferulus martini**
		?*Paratendipes* sp V12*
		Polypedilum sp V3*
		Polypedilum sp V33*
		Rheotanytarsus sp 1
		Rheotanytarsus sp 2
		Riethia sp V4*
		Riethia sp V5*
		Stempellina sp*
		Limnophyes sp*
		?*Limnophyes* sp 1
		?*Limnophyes* sp 2*
		Orthocladinae sp SW23
		Orthocladinae sp SW28
		Orthocladinae sp SW29
		Orthocladinae sp SW30
		Orthocladinae sp VCD2
		*Cricotopus annuliventris**
		*Stictocladius uniserialis**
		Stictocladius sp SW32
		?*Parakiefferiella* sp *
		Thienemanniella sp *
		Ablabesmyia sp*
		Macropelopia dalyupensis
		*Paramerina levidensis**
		Paramerina sp*
		Procladius sp*

Table 2

Macroinvertebrates recorded in Carey Brook in the karri forest by I.O. Growns but not found by Pusey and Edward (1990a).

Taxonomic group	Common name	Species
Mollusca	Snails and mussels	*Westralunio carteri*
Hydracarina	Watermites	*Piona* sp
Ostracoda	-	*Candonocypris novaezelandiae* *Bennelongia ?barangaroo*
Isopoda	-	*Hyperdesipus ?plumosis*
Zygoptera	Damselflies	*Argiolestes pusillus* *A. minimus*
Anisoptera	Dragonflies	*Acanthaeschna anacantha* *Austrogomphus lateralis* *Synthemis cyantincta* *Hemigomphus armiger* *Lathrocordulia metallica*
Megaloptera	Alderflies	*Archichauliodes* sp
Ephemeroptera	Mayflies	*Tasmanocoenis* sp *Baetis soror* *Neboissophlebia* sp undescribed genus Q undescribed genus R
Trichoptera	Caddisflies	*Taschorema pallescens* *Apilsochorema urdalum* *Ecnomina sentosa* s.1. *E. scindens* s.1. *Diplectrona* sp *Smicrophylax australis* *Adectophylax ?volutus* *Hydrobiosella* sp (MMBW sp 16) *Kosrheithrus boorarus*
Tanyderidae	-	*Radinoderus ?occidentalis* *Eutanyderus* sp
Pelecorhynchidae		*Pelecorhynchus* sp

Table 2 *(continued)*

Taxonomic group	Common name	Species
Psychodidae	Mothflies	two species
Tipulidae	Craneflies	eleven species
Empididae	-	two species
Athericidae	-	one species
Ceratopogonidae	Biting midges	four species
Thaumaleidae	-	three species
Tabanidae	March flies	two species

Table 3

Numbers of species in different taxonomic groups in streams of northern jarrah forest (Bunn *et al.* 1986) compared with preliminary data from karri forest streams (Tables 1 and 2).

Taxonomic group	Common name	No of species	
		Jarrah	Karri
Cnidaria	Hydras	1	-
Platyhelminthes	Flatworms	2	-
Nematoda	Roundworms	1	-
Nematomorpha	Gordian worms	1	-
Mollusca	Snails	3	4
Annelida	Worms and leeches	1	1
Hydracarina	Water mites	1	12
Ostracoda	-	1	12
Copepoda	-	2	5
Syncarida	-	1	-
Isopoda	-	1	3
Amphipoda	-	2	4
Decapoda	Crayfish	3	1
Ephemeroptera	Mayflies	6	7
Zygoptera	Damselflies	2	3
Anisoptera	Dragonflies	9	5
Plecoptera	Stoneflies	3	3
Hemiptera	Bugs	2	3
Megaloptera	Alderflies	1	1
Tipulidae	Craneflies	5	12
Psychodidae	Mothflies	2	2
Culicidae	Mosquitoes	1	2
Chironomidae	Midges	33	36
Ceratopogonidae	Biting midges	5	7
Simuliidae	Blackflies	8	4
Thaumaleidae	-	2	3
Tabanidae	Marchflies	1	2
Athericidae	-	1	1
Dolichopodidae	-	1	-
Empididae	-	3	3
Trichoptera	Caddisflies	26	27
Coleoptera	Beetles	14	12

Cherax tenuimanus is particularly important because marron fishing constitutes a major recreational pursuit in south-western Australia and the larger rivers in the karri forest are intensively fished when the season is open (Morrissy 1978; Morrissy *et al.* 1984). In the 1970s the biomass of *Cherax tenuimanus* in the lower reaches of the Warren River exceeded 600 kg ha^{-1} (N.M. Morrissy[5] personal communication), making it the dominant animal species in the system. However, there has been a dramatic decline in numbers of *Cherax tenuimanus* throughout the species' range in recent years. This led to the closure of the marron fishery for the 1987/88 season (Fisheries Department 1988). The cause of the decline appears to be over-fishing and the increasing eutrophication of rivers (Morrissy *et al.* 1984), combined with a series of dry years.

Nine species of native fish are thought to occur in karri forest streams, eight of which are endemic to south-western Australia (Table 4; Christensen 1982). *Lepidogalaxias salamandroides* is of considerable scientific interest; Rosen (1974) claims it is the sole southern hemisphere representative of the esocoid group, which includes northern hemisphere pike.

In addition to the other nine species, *Favinogobius suppositus* has been recorded in Big Brook (L.J. Pen[6] personal communication) although that is far removed from its recognized habitat (Allen 1982). Both *Favinogobius suppositus* and *Pseudogobius olorum* occur in the estuaries of rivers passing through the karri forest and extend into the lower reaches of the rivers, as well as occurring in coastal lakes. The estuarine *Leptatherina wallacei* has moved upstream into the Avon River from the Swan Estuary and, because it occurs in south-coast estuaries, may do the same in rivers passing through the karri forest as they become increasingly saline (L.J. Pen personal communication).

Four species of introduced fish have been recorded in karri forest streams, namely *Salmo trutta, Onchorhynchus mikiss, Perca fluviatilis* and *Gambusia affinis* (I.C. Potter[7] and L.J. Pen personal communication) although *Gambusia affinis* is rarely found. It is possible that *Carassius auratus* and *Carassius carassius* also occur but they have not been recorded.

Feeding ecology

The aquatic invertebrates inhabiting streams and rivers can be divided into four more-or-less distinct groups on the basis of their feeding ecology (Cummins and Klug 1979). Analysis of stream faunas according to the number of species in each feeding group has frequently been used to indicate the relative importance of different kinds of energy input into the stream or as an indicator of stream health (e.g. Gurtz and Wallace 1984; Bunn 1986b). This work is under-pinned by an extensive body of theory, best summarized in the River Continuum Concept (Vannote *et al.* 1980; Minshall *et al.* 1985). The feeding groups are (Cummins and Klug 1979):

Shredders - (e.g. amphipods) feed on CPOM. The shredder/CPOM system is a complex one and is best

Table 4

Species of native fish in karri forest streams, their conservation status and habitat preferences in south-western Australia.

Scientific name	Common name	Preferred habitat type[a]	Main distribution in woodchip area[b]	Range[c]	Status[b]
Bostockia porosa	Nightfish	stream	all habitats	south-west	common
Edelia vittata	Western pigmy perch	stream	forest, especially karri	south-west	common
Galaxiella nigrostriata	Black-stripe minnow	pond	?flats	?inland from Albany	rare
Galaxias occidentalis	Western minnow	stream	?forest, especially jarrah	Perth to Albany	mod. common
Galaxiella munda	Mud minnow	pond	jarrah and flats	Perth to Albany	common
Lepidogalaxias salamandroides	Salamander fish	pond	jarrah and flats	Augusta to Albany	common
Nannatherina balstoni	Balston's pigmy perch	stream	?all habitats	Augusta to Albany	rare
Geotria australis	Pouched lamprey	stream	?all habitats	Perth to Albany	common[d]
Tandanus bostocki	Freshwater cobbler	river[d]	-	coastal south-west	?common[d]

[a] G.R. Allen personal communication

[b] Christensen (1982)

[c] Allen (1982)

[d] I.C. Potter and L.J. Pen personal communication

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[6] L.J. Pen, Western Australian Waterways Commission.

[7] Prof. I.C. Potter, School of Biological and Environmental Sciences, Murdoch University.

explained with reference to leaf material. The same principles apply to other types of CPOM but processing times are much longer. When leaves fall into the water (or, in the cases of temporary streams, are first inundated) there is a fairly rapid initial loss of weight as the leaves are leached, i.e. DOM is produced. This is followed by a period when leaf weight remains fairly constant and the leaves are colonized by bacteria and hyphomycete fungi. These microbes transform CPOM to FPOM and DOM through their own metabolism and significantly improve the digestibility of the leaves for shredders. The major production of FPOM occurs, however, as CPOM is broken down by shredders eating it and passing the digested fragments out as FPOM in their faeces. Physical abrasion on the stream bed also results in CPOM being broken down into FPOM.

Collectors - feed on FPOM. Collector-gatherers, such as many chironomid and ephemeropteran larvae, feed on sediment-related FPOM whereas collector-filterers, of which simuliid and net-spinning trichopteran larvae are examples, feed on FPOM carried in the water column. Collectors feed on FPOM from a variety of sources: direct input from the stream headwater, CPOM broken down by shredders or abrasion, DOM that has flocculated or which decomposing microbes have metabolized and the faeces of other invertebrates.

Scrapers - (e.g. limpets and some ephemeropteran larvae) graze periphyton and other food adhering to solid surfaces. Scrapers are more common in autotrophic systems, often in areas of faster flowing water.

Predators - (e.g. zygopteran larvae) capture live prey.

Some species are polytrophic and occupy several of the above feeding categories. The *Cherax* species occurring in the karri forest are probably primarily shredders but also feed on live plants (particularly plant roots) and other animals, including their young (N.M. Morrissy personal communication).

Although they are not usually fitted into the above feeding scheme, fish are either predators or scrapers that also feed on macrophytes (Hynes 1970). All the native fish, except *Geotria australis*, occurring in karri forest streams are predators, feeding mainly on aquatic insects and insect larvae although *Galaxias occidentalis* feeds extensively on terrestrial insects (Pen and Potter 1991b) and *Edelia vittata* and *Nannatherina balstoni* occasionally eat juvenile fish. The composition of the diet of *Lepidogalaxias salamandroides* and, presumably, other species shows changes during the year (G.R. Allen[8] personal communication). Larval *Geotria australis* feed on algae and on bacteria associated with decaying organic matter; adults do not feed while in freshwater (Potter *et al.* 1986).

In an analysis of the functional organization of the invertebrate communities in jarrah forest streams in south-western Australia, Bunn (1986b) found that low-order streams were dominated by detritivores, especially collectors, which confirmed that the streams are heterotrophic. Shredders occurred but were less common than predicted by the River Continuum Concept (Vannote *et al.* 1980), probably because of the refractile nature of eucalypt leaves. [In recent years the general applicability of the River Continuum Concept, especially to Australian streams, has been questioned (e.g. Lake and Barmuta 1986).] As expected, collector-filterers were more common in winter, when FPOM is in suspension, than in summer when flow rates are low. Less predictably scrapers, which feed on algae, were more common in winter than summer. The reason for this was unclear but it suggested that primary productivity is never high in these small, closed-canopy streams. Lebel (1987) confirmed Bunn's results.

There have been no studies of organic matter processing in karri forest streams or the major energy inputs so it is impossible to do more than speculate about the functional organization of karri forest streams. However, it appears reasonably certain that the major source of carbon in low-order streams is allochthonous and preliminary indications from the work of I.O. Growns are that shredders are an important component of the fauna of these streams (see **Energy Sources**).

Microhabitats

Adults of most insect groups occurring in streams are terrestrial. They usually live in the riparian zone beside the stream and are dependent on riparian vegetation for their habitat (e.g. Riek 1970a,b).

Aquatic insect larvae and other aquatic invertebrates are directly dependent on stream habitat, utilizing four main substrata within the stream:

(1) rocks/boulders/pebbles;

(2) gravel/sand/silt;

(3) organic material (wood, leaves etc);

(4) aquatic plants.

Selection of particular substrata by individual species is influenced by feeding requirements, water velocity, biological interactions and other factors (Hynes 1970; Mackay and Wiggins 1979; Pinder 1986).

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[8] Dr G.R. Allen, Department of Ichthyology, Western Australian Museum.

Preliminary work by I.O. Growns (personal communication) has shown that in first and second-order tributaries of Carey Brook in the karri forest, sand/silt substrata are populated by large numbers of trichopterans and amphipods. Plecopterans and ephemeropterans occur but in smaller numbers. The number of animals in the mid-sections of Carey Brook, among the boulder/pebble riffles, is extremely variable. In some areas trichopterans, simuliids and ephemeropterans are common. Plectoperan abundance seems to be related to season, being high in spring and low in summer. Chironomids comprise approximately 50 per cent of the invertebrate population at any location. In areas of accumulated leaf debris ostracods and calanoid and harpacticoid copepods occur as well as amphipods.

Cherax tenuimanus adults live on the bottom in deeper sections of rivers, which usually have sand substrata. Juveniles live among the litter in shallow water, to escape predation by adults, in shaded sections of rivers where poor visibility prevents predation by birds (N.M. Morrissy personal communication).

Although a few species of fish are confined to particular types of substratum, most fish occur independently of substratum except when spawning. Depth of water and velocity of current are more important in determining fish distribution (Hynes 1970). Shelter is also important to most species of fish, largely because it provides refuge from the current, and they tend to congregate where log-jams, macrophytes or changes in stream contour provide this (Hynes 1970). The fish species in karri forest streams probably require shelter, especially *Edelia vittata* (Hutchinson 1991) and *Nannatherina balstoni*, but species that occur mostly in ponds such as *Lepidogalaxias salamandroides* and *Galaxiella nigrostriata* are frequently found in areas with a clear bottom (G.R. Allen personal communication).

Life histories

There has been comparatively little work published on life histories of aquatic invertebrates in Australia and almost nothing on species occurring in streams of the south-west other than Edward (1986) and Bunn (1988b). The work published on plecopterans, ephemeropterans and trichopterans in eastern Australia showed that a tremendous array of life cycles can occur, with some species having larval/nymphal stages of three or four years duration while others can have three generations a year. Nevertheless the most common pattern is a univoltine one (i.e. one generation per year) (Hynes and Hynes 1975; Marchant *et al.* 1984; Yule 1985; Campbell 1986; Dean and Cartwright 1987). Rates of development are strongly influenced by temperature and, it seems, to a lesser extent by food (Campbell 1986). Different temperatures can alter the degree of synchrony in emergence and, in some cases, can affect the number of generations per year. The very small amount of work done in jarrah or karri forest streams suggests that a number of the animals occurring there, and particularly the Gondwanaland relicts, are probably univoltine (Bunn 1988b; D.H.D. Edward personal communication).

The temporary nature of many first-order streams in the karri forest means that the species inhabiting them must have special adaptations in terms of life histories or habitat preferences that enable populations to be maintained long-term (Towns 1985; Boulton and Suter 1986). Possible adaptations in life histories are:

(1) having a drought-resistant stage to survive the period when the stream is dry;

(2) recolonizing the stream from nearby pools (on the same watercourse) or adjacent permanent streams after above-ground flow recommences. This requires fairly strong powers of dispersal (e.g. a flying adult stage) to recolonize from adjacent streams, or tolerance of low oxygen levels and other adaptations to still water if recolonization is to occur from pools in the creek system;

(3) rapid larval growth during periods of stream flow.

Alternatively, animals can survive dry periods by moving down into the stream bed to where water is still present. Koonacs and, to a lesser extent, gilgies do this by burrowing (N.M. Morrissy personal communication) and D.H.D. Edward (personal communication) suggests this is a common phenomenon amongst macroinvertebrates in jarrah (and probably karri) forest streams although it has not been properly documented. The best data documenting this type of movement come from the northern hemisphere (e.g. Williams and Hynes 1976, 1977).

Most species of fish live several years. Often they are sedentary in the non-breeding season although some species exhibit seasonal movements related to water depth or other environmental factors. Many species migrate to breed. In some cases fish that spend their adult life at sea or in estuaries move into rivers to lay eggs, the larvae from which spend the early development period in rivers before moving out to the sea. Many lake-dwelling species also spawn in rivers. Frequently river-dwelling fish move upstream into low-order creeks or onto flood-plains to breed. Timing of the breeding season is determined by photoperiod, water temperature or flow rate (Hynes 1970).

Although there has been little work on fish in the karri forest, there has been a recent spate of studies in other habitats on the biology of fish that occur in karri forest streams. *Lepidogalaxias salamandroides* and *Geotria australis* are the most intensively studied species. Allen and Berra (1989) have shown that *L. salamandroides* lives three or four years, has high annual mortality and breeds any time of the year that there is sufficient water. McDowall and Pusey (1983) reported that adults dig a tubular burrow and aestivate in damp mud when water dries out in the pools they are occupying. They also stated that eggs are laid in spring and are probably drought-resistant; Allen and Berra (1989) reported hatching in spring and implied that young fish aestivate as pools dry out. Males become sexually mature at the end of their first year of life, in females maturity is delayed until the second year.

Geotria australis has a more complicated life history (Potter *et al.* 1986). Spawning in south-western Australia (e.g. Donnelly River) occurs in October or November, the young then spend four or five years as larvae. Metamorphosis begins in January or February (just over four years after spawning) and is completed by the time downstream migration occurs in July or August. The downstream migration into the ocean coincides with increased discharge rates in rivers.

After spending a considerable time in the ocean, where they feed on fish, adults return to freshwater to spawn. They enter the Donnelly and Warren Rivers in June or July and spawn in October or November of the following year.

Galaxias occidentalis, Bostockia porosa and *Edelia vittata* have been studied in the Collie River, which is in the jarrah forest (Pen and Potter 1990, 1991a,b). All three species spend summer and autumn in the main branch of the Collie River. They breed in late winter/early spring after undertaking an upstream migration during winter. Sexually maturing *Galaxias occidentalis* occur predominantly in stream headwaters and low-order creeks but *Bostockia porosa* and *Edelia vittata* occur in significant numbers at this time in floodwaters adjacent to the river.

Galaxias occidentalis and *Edelia vittata* populations consist predominantly of fish in their first year of life although they can survive up to three years. Sexual maturity is reached at the end of the first year. *Bostockia porosa* is longer-lived, with a small proportion of the population living beyond three years (maximum life span is at least six years), and females do not mature until the end of their second year although some males mature in the first year.

Pusey and Edward (1990b) provide some information about the life cycles of *Galaxiella munda, Galaxiella nigrostriata* and *Nannatherina balstoni.* However, there are substantial gaps in knowledge of the latter two species; L.J. Pen and I.C. Potter (unpublished data) have fairly comprehensive data on *Galaxiella munda.* M.J. Hutchinson[9] (personal communication) has recently observed the spawning behaviour of *Tandanus bostocki* but there is little known about the life history of this species.

Forest Management

Timber harvesting

There are numerous overseas studies showing that land disturbance in general, including clearing for agriculture, timber harvesting, road construction and wildfire, can cause substantial changes in the composition of fish and stream invertebrate communities (e.g. Graynoth 1979; Newbold *et al.* 1980; Gurtz and Wallace 1984; Wallace and Gurtz 1986; see Blackie *et al.* 1980; Doeg and Koehn 1990b for bibliographies). Whether timber harvesting and associated road construction cause changes in the biota of Australian streams is less clear. By analogy with overseas situations, changes have been predicted (Michaelis 1984; Campbell and Doeg 1989; Doeg and Koehn 1990a,b) but results are available from only three studies.

Robinson (1977) and Richardson (1985) both suggested that timber harvesting resulted in changes in streams communities in eastern Australia. Doeg and Koehn (1990a) have pointed out methodological problems in both Robinson's and Richardson's work but concluded that the differences observed by Richardson above and below a logged section of stream and a road crossing nine months after logging ceased were probably real. This is in contrast to what was implied by the work of Cornish (1980) in the same area, who found that logging had little effect on the quality of drinking water. The studies of Richardson and Cornish suggest that methods traditionally used to monitor the effect of logging on water quality *per se* are inadequate to predict what effect there will be on stream fauna.

A recently completed study in the karri forest of Western Australia showed that timber harvesting without stream buffers resulted in changes to community structure (but not species richness or abundance) that lasted at least eight years after harvesting ceased (Growns and Davis 1991). The community structure of a stream with a buffer 100 m

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[9] M.J. Hutchinson, Department of Geography, The University of Western Australia.

wide was similar to unlogged control streams; Growns and Davis (1991) concluded that a 100 m wide buffer prevented community changes.

There are nine ways by which timber harvesting and associated road construction might cause changes to the aquatic communities of karri forests streams. In all cases changes can be prevented by retaining adequate buffer strips of undisturbed vegetation and by careful design and construction of road crossings over streams.

(1) Most sediment originates from roads and snig tracks, especially where they cross streams, and landings constructed as part of the timber harvesting operation (Melbourne and Metropolitan Board of Works 1980; Campbell and Doeg 1989). Leaving buffer strips of intact vegetation on both sides of streams and constructing stream crossings carefully substantially reduces the amount of sediment entering streams as a result of timber harvesting (Clinnick 1985).

Most sediment transport occurs during very short periods after rain (Gilmour 1971; Weston *et al.* 1983). Campbell and Doeg (1989) showed that many studies of water quality in Australian streams have under-estimated sediment transport because peak flows were not sampled. This, together with the fact that sediment levels are largely influenced by topography and soil type as well as intensity of rainfall events (Olive and Walker 1982), makes it difficult to compare the severity of impacts in different studies.

There are two types of sediment that can affect aquatic fauna - *suspended* and *deposited.* Whiteley (1978) observed increased suspended sediment levels during wet weather in most karri forest streams he sampled after harvesting with buffers. The levels of flow-weighted suspended sediment levels increased six-fold in a March Road stream in the karri forest after clear-felling without buffers (Borg *et al.* 1987a). They reached a maximum concentration of about 550 mg L^{-1} during heavy rain (H. Borg[10] personal communication), which is similar to the maximum level of 860 mg L^{-1} recorded by Graynoth (1979) in New Zealand, where increased sediment had pronounced effects on the fauna.

Increased levels of suspended sediment can clog the gills of some animals and kill them directly; it also increases rate of invertebrate drift. Doeg and Milledge (1991) provided preliminary data suggesting the threshold at which suspended sediment affects some aquatic invertebrates in eastern Australia is <250 mg L^{-1}. The ability of different species to tolerate sediment varies greatly, however, and an indication of this variability and/or the extent of natural disasters is provided by the fact that the concentration of suspended sediment can exceed 100 000 mg L^{-1} in streams flowing through areas prone to erosion after wildfire and heavy rain (see **Fire**).

Deposited sediment can carpet the stream bed (reducing the availability of organic matter and making conditions anoxic) or can fill the interstitial spaces in which many invertebrates live (Cordone and Kelly 1961; Chutter 1969; Doeg *et al.* 1987; Bunn 1988b). Bunn *et al.* (1986) compared the faunas of 'undisturbed' streams and those in which sediment had been deposited over the stream bed as a result of bauxite mining operations in the Western Australian jarrah forest. They found little difference apart from the occurrence of a silt-tolerant species of ephemeropteran in sedimented streams. However, subsequently Bunn (1988b) found the breakdown of CPOM by invertebrates was virtually eliminated in the sedimented streams. There were substantial differences between the leaf litter communities of sedimented and undisturbed streams: sedimented streams contained more animals but they were mostly predators and very few shredders were present.

Richardson (1985) attributed the changes she had observed in the invertebrate fauna downstream from timber harvesting and a logging road across a stream in New South Wales to the effect of increased sedimentation. Elevated sediment levels were one of the factors that appeared important in changing community composition in the study by Growns and Davis (1991) in the karri forest.

(2) Increased concentration of salts in streams has been shown to affect community composition in North American and New Zealand streams (Bormann *et al.* 1974; Graynoth 1979; Lemly 1982). The extent and duration of increases in salinity in Western Australian forest streams after timber harvesting depends on several factors of which rainfall and buffer width are most important (Borg *et al.* 1987b). In the most extreme cases without buffers in the karri forest, salinities will take about ten years after timber harvesting ceases to return to pre-logging values. Buffers largely prevent increases in salinity (Borg *et al.* 1987b).

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[10] H. Borg, Water Authority of Western Australia.

Harvesting without a buffer, however, caused salinities during periods of low flow-rate in a March Road creek in the karri forest to increase more than two-fold (from 700-1500 mg L^{-1} for a month, Steering Committee for Research on Land Use and Water Supply 1987). Growns and Davis (1991) observed changes in the community composition of March Road, and another stream where harvesting occurred without a buffer, and concluded that salinity was one of the factors responsible for the changes.

Hart *et al.* (1990, 1991) suggested that adverse biological effects occur in Australian streams if salinities increase beyond 1000 mg L^{-1}. Robson (1990) noted changes in the invertebrate fauna of streams in wandoo forest of Western Australia as a result of salinities increasing to 3000-5000 mg L^{-1}.

(3) Elevated water temperature owing to lack of shading after harvesting affects the life cycles of many animals and, thus, community composition (Brown and Krygier 1970; Lee and Samuel 1976; Rishel *et al.* 1982). Because of the comparatively high temperatures in south-western Australia, it is likely that harvesting without buffers would elevate water temperature enough to disrupt life cycles of some species until the same level of shading is achieved by regrowth (see Hynes and Hynes 1975; Campbell 1986; Bunn 1988b). The increase in temperature that occurs when harvesting without buffers has not been measured in Western Australia.

Increased temperatures can be prevented by leaving buffer strips of undisturbed vegetation.

(4) Increased input of FPOM (or fine sediment) can lead to eutrophication in streams, with concomitant effects on the fauna (Graynoth 1979; Bunn 1986a). However, adequate buffers prevent elevated input of FPOM into streams.

Borg *et al.* (1988) observed increases in FPOM in the karri forest streams in the Sutton and Poole blocks after harvesting without a buffer. In the subsequent two years algal blooms developed in the Sutton block stream. This was probably the result of increased insolation of the stream and, perhaps, nutrient enrichment as a result of the extra input of organic matter. It is possible that eutrophication *per se* would affect karri stream communities (see Lemly 1982), in addition to the direct effects of increased insolation on temperature and, hence, productivity.

Elevated FPOM eight years after harvesting was one of the differences between logged and unlogged streams in the Sutton and Poole blocks studied by Growns and Davis (1991) where differences in community compositions occurred. The lower nitrogen levels in logged streams, however, suggested that the differences between streams were not the result of nutrient enrichment.

(5) Deposition of logging debris in streams can result in substantial changes to stream profiles, thus affecting fish and some aquatic invertebrates (Graynoth 1979; Winterbourn 1986; Campbell and Doeg 1989). However, retention of buffer strips ensures that logging debris does not enter streams.

Borg *et al.* (1988) observed significant changes in the profile of the stream channel when logging was conducted on the banks of the Sutton block stream in the karri forest. Graynoth (1979) showed that this affected the biota of streams in New Zealand.

(6) Short-term alteration in the pattern of input of allochthonous material and increased primary production (as a result of opening the canopy) may lead to changes in community composition (Lyford and Gregory 1975) although this is not the case in studies undertaken in New Zealand (Winterbourn 1986). Most studies have found increased use of periphyton by invertebrates after timber harvesting. In spite of theoretical expectations, there are no data indicating that changed patterns of allochthonous input have affected invertebrates (Campbell and Doeg 1989). The likelihood of these effects in the karri forest is unknown but adequate buffer strips circumvent the problem by preventing increased primary production and changes in patterns of allochthonous input.

(7) Long-term reduction in litter input from riparian vegetation after timber harvesting and regeneration may affect aquatic communities although adequate stream buffers should prevent a reduction of litter input.

There are two components of litter used by aquatic animals - *leaf litter* and *branches or logs*. For many years after regeneration forests in North America release less leaf litter into streams and the litter released has a different species composition from that released by mature forest (Webster and Waide 1982; Webster *et al.* 1987). This leaf litter provides an energy source for invertebrates inhabiting streams but leaves of some plant species that are abundant in regenerating Australian forests, such as *Acacia,* are not palatable to stream invertebrates

(Campbell and Doeg 1989). Overseas studies suggest that the reduction of leaf litter input may result in a depauperate stream community (Haefner and Wallace 1981).

Although a lot of branches and logs may enter streams during harvesting, with the consequences discussed in (5) above, in the longer term the quantity of branches and logs entering streams is reduced after timber harvesting. Golladay *et al.* (1987) suggested the number of debris dams may be depressed for 100-400 years after harvesting and regeneration in North America. There is evidence to suggest the reduction of debris reduces fish and invertebrate populations (Elliott 1986; Hutchinson 1991) through loss of habitat and cover.

(8) Even if CPOM is the major energy source in karri forest streams the input of FPOM and DOM from the swampy stream headwaters will still be important (see **Energy Sources**). Harvesting areas adjacent to headwaters without buffers may result in community changes in downstream areas because of different organic matter inputs (as well as increased sediment loads), even though the downstream reaches are protected by buffer strips.

(9) Poor road design where roads cross streams can result in barriers to the movement of fish. However, roads are only one of a number of barriers to fish movement in south-western Australia; others include farm dams on streams, gauging stations, debris pushed into streams during land clearing and natural obstructions. Cumulatively, the man-made barriers may lead to reductions in populations of native fish through limiting access to potential breeding habitat, preventing dispersal between habitats and trapping fish in receding pools during dry periods (L.J. Pen and M.J. Hutchinson personal communication).

Like many roads serving other purposes in the karri forest, the stream crossings of some logging roads cannot be easily negotiated by native fish. Appropriate design specifications are now available for road crossings and gauging stations (Aquatic Research Laboratory 1991; L.J. Pen personal communication); farm dams and careless land clearing remain problems.

As already indicated, adverse effects of timber harvesting and associated road construction can be prevented by leaving buffers of undisturbed riparian vegetation along streams. Clinnick (1985) reviewed published data and concluded that for Australian timber harvesting operations a buffer of 30 m either side of the stream should be left to ensure acceptable water quality and minimal disturbance of the stream ecosystem. Clinnick (1985) regarded a buffer of at least 20 m width along temporary streams and stream headwaters as essential to achieve this protection.

So far, the adequacy of the buffer widths advocated by Clinnick has not been investigated experimentally with respect to invertebrates and fish (see Campbell and Doeg 1989). However, Growns and Davis (1991) found that wider buffers (100 m) along karri forest streams provided adequate protection for the aquatic community and commented that any buffer width that maintained water quality would probably be adequate. Borg *et al.* (1988) found that 50 m wide buffers were sufficient to maintain water quality in karri forest streams, although Whiteley's (1978) study suggested wider buffers (or tighter controls over harvesting operations including, perhaps, better road design) may be required. Correct implementation of harvesting prescriptions to prevent breaches of buffer strips is probably a more critical issue in stream protection than the exact width of buffer used. At present buffer strips of 50-200 m either side of the watercourse are used along all rivers and many streams in the karri forest and it is likely that this system will be extended (R.J. Underwood[11] personal communication).

Fire

There is little information available about the effect of fire on stream invertebrates (see Campbell and Doeg 1989) but the principles that apply to timber harvesting should also apply to fire when similar levels of disturbance result.

Low-intensity fires, such as prescribed fuel-reduction burns, have little effect on soil-water infiltration rates or other parameters likely to affect water quality in streams (Gilmour and Cheney 1968) but wildfires can increase sediment levels, salinity, nutrient levels and the amount of organic material in streams (Shea *et al.* 1981; Olive and Walker 1982; Campbell and Doeg 1989). The extent of increases in the above parameters depends on fire intensity, nature of the catchment, period between fire and the first significant rainfall event, and the intensity of the rainfall event (Chessman 1986; Campbell and Doeg 1989).

Increased accession of sediment to streams is probably the effect of fire that has most impact on stream invertebrate and fish communities although there is often a pronounced increase in stream discharge as well. Seven months after wildfire in the

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[11] R.J. Underwood, Department of Conservation and Land Management.

Snowy Mountains, New South Wales, sediment concentrations of 112 000 mg L^{-1} and 143 000 mg L^{-1} were measured in creeks that had highest recorded pre-fire maxima of 334 mg L^{-1} and 7052 mg L^{-1}, respectively (Brown 1972). Chessman (1986) found sediment concentrations of 2300 mg L^{-1} in the Cann River after the Ash Wednesday fires in Victoria. Lower concentrations (47-283 mg L^{-1}) have been recorded elsewhere (Midgley 1973; Burget *et al.* 1980, 1981; Chessman 1986).

Depending largely on the rate of revegetation, water quality often improves rapidly after wildfire although sediment will remain in the streambed. Chessman (1986) noted that water quality had begun improving four months after the Ash Wednesday fires. In contrast, Burgess *et al.* (1980, 1981) reported that wildfire near Eden, New South Wales, caused larger changes of greater duration in sediment load than did clear-felling. The greatest increases in sediment loads were caused by timber harvesting followed by high-intensity fire (Burgess *et al.* 1981).

CURRENT RESEARCH

Research currently underway in karri forest streams can best be summarized under the headings of **Fauna** and **Impacts of timber harvesting.** We have also summarized relevant studies in the jarrah forest and other parts of south-western Australia. Doeg and Koehn (1990a) have recently summarized current forest stream research in Australia on a State-by-State basis.

Fauna

Fish

Three institutions are involved in fish research in the karri forest.

At the University of Western Australia, B.J. Pusey recently completed a PhD in the Zoology Department on the biology and life history of *Lepidogalaxias salamandroides* and other native fish in the flats on the southern edge of the karri forest (Pusey and Edward 1990a,b). M.J. Hutchinson has just finished a PhD in the Geography Department, studying the factors affecting fish distribution in the Murray River, including the effect of the introduced trout (*Salmo trutta*) and *Perca fluviatilis* and the effect of man-made barriers, such as gauging weirs and roads (Hutchinson 1991). The Aquatic Research Laboratory in the Zoology Department, led by D.H.D. Edward, is carrying out some research into the biology of fish in the jarrah forest as part of their extensive monitoring program for the Western Australian Water Authority (Aquatic Research Laboratory 1988, 1991; Pusey *et al.* 1989).

At Murdoch University there is a strong research group studying fish in the School of Biological and Environmental Sciences, although the only work being done in the karri forest is a study by L.J. Pen on the effect of dams and weirs on fish migration being conducted at Big Brook for the Water Authority (Pen *et al.* 1988) and research being conducted on lampreys (*Geotria australis*) by I.C. Potter and colleagues (Potter *et al.* 1986). However, the results of work by Pen and Potter (1990, 1991a,b) on the biology of fish in the Collie River are relevant to management of stream fauna throughout the south-west.

G. Allen and T. Berra of the Western Australian Museum are carrying out inventories and taxonomic research throughout south-western Australia, including the karri forest, and have recently completed a detailed study of the life history of *Lepidogalaxias salamandroides* (Allen and Berra 1989).

Aquatic invertebrates

A PhD study in the karri forest by I.O. Growns of the School of Biological and Environmental Science, Murdoch University is currently being written up (discussed under **Impacts of timber harvesting**).

The Fisheries Department has a general research and monitoring program for *Cherax tenuimanus*, led by N.M. Morrissy, but no specific research is being done in the karri forest.

A considerable amount of research has been undertaken recently on the invertebrates of jarrah forest streams (Bunn 1986a,b, 1988a,b,c; Bunn *et al.* 1986; Storey and Edward 1989; Storey *et al.* 1990, 1991).

Impacts of timber harvesting

Growns is about to submit his PhD. He has addressed two questions: (1) whether timber harvesting affects aquatic invertebrate communities in karri forest streams, and (2) whether the communities differ between first and higher-order streams. He worked on Carey Brook and the Department of Conservation and Land Management altered its timber harvesting program in the area to provide appropriate logged and unlogged areas for short-term studies. Some of the paired catchments studied by the Steering Committee for Research on Land Use and Water Supply (1987) were also examined to provide a long-term perspective (Growns and Davis 1991).

Some of the research being conducted in the jarrah forest by the Aquatic Research Laboratory at the University of Western Australia will also provide useful information about the effect of logging on aquatic invertebrates. This includes PhD studies by P. Davies on community structure and function in relation to autotrophy/heterotrophy and by L. Lebel on trichopterans, as well as studies cited under the previous **Aquatic invertebrates** section.

HIGH PRIORITY ADDITIONAL RESEARCH REQUIREMENTS

The five research topics involving karri forest streams that should receive highest priority are:

Inventory of aquatic invertebrates - the first step in devising a conservation strategy for stream fauna in the karri forest is finding out what animals occur where and determining their conservation status. There is a need for broad-scale surveys to document the fauna of karri forest streams and to establish which species are endemic, how restricted their distribution is and which parts of the stream system they inhabit. It is essential that animals are named in this type of work; voucher collections have very limited value and, hence, the work must have a taxonomic component.

Impact of logging on aquatic communities - just as important as undertaking an inventory, is the need to quantify further the impact of timber harvesting and subsequent regeneration on the biota of karri forests streams. This study could be undertaken in conjunction with an inventory. Growns and Davis (1991) have shown that changes in community composition persist at least eight years when areas adjacent to streams are harvested without a buffer. The main issues to be addressed are: (1) whether Growns and Davis' (1991) results are typical of karri forest streams or reflect peculiarities of their sites, (2) which species, if any, disappear from streams as a result of harvesting adjacent forest, (3) how long changes in community composition persist, and (4) how wide buffers must be to prevent given levels of change. The results of the inventory could be used to determine which streams require full protection and which streams could be allowed to have a period of perturbation. Buffer widths could be set accordingly.

Sources of energy input - management of karri forest streams is dependent on understanding how they function. A study of energy flow through the stream system, in particular identifying the major sources of energy input and documenting how the community is structured around energy transfer through the system, would provide much of the required information. Direct studies of the effect of timber harvesting and regeneration are essential and have the advantage of producing results fairly quickly but usually provide limited information about why particular results are obtained. Longer-term studies, in conjunction with direct studies of the effect of harvesting, will provide a better scientific basis for the design of buffer systems, harvesting prescriptions and stream management guidelines.

Structure and composition of stream substrata - as noted under **Forest Management**, input of sediment in streams as a result of land disturbance occurs as occasional short-term events. The effects on stream fauna will largely depend on two sets of factors:

(1) whether fine sediment penetrates the stream bed, how long it remains and whether it changes the structure or composition of the bed.

(2) whether the stream fauna is dependent in some way on interstitial features of the stream bed such as:

  (a) interstitial spaces free of fine inorganic sediment, which certain species may move through at a stage in their life history or use as refugia in dry periods;

  (b) high organic: inorganic ratio in fine material to act as a food source for collectors-gatherers.

The extent of sediment input and its effect on the stream fauna could be addressed comparatively simply using established methods of collecting cores from stream beds and by subsequent analysis of the fauna present at different depths in relation to bed structure and composition.

Life cycles and spawning in fish - management of fish species, like other animals, requires knowledge of life cycles. This information is currently lacking for *Galaxiella nigrostriata, Tandanus bostocki,* and *Nannatherina balstoni* and should be collected.

Even for species with known life-cycles, information about spawning behaviour, egg development and hatching are not available and in most cases the spawning areas are not known. Because recruitment is such a critical phase of the life cycle it is important that there is a systematic study of the spawning areas and early development of the native fish of the karri forest.

ACKNOWLEDGMENTS

We wish to thank J.R. Bartle, M.J. Hutchinson, L. Lebel, L.J. Pen, Prof. I.C. Potter and Drs B. Knott, N.M. Morrissy, G.R. Allen, T. Berra and J.A. Davis and, particularly, I.O. Growns and Dr D.H.D. Edward for providing much useful information and assistance in compiling this review.

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