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Acari (Mite) Abundance And Diversity In Soil And Litter Layers Of Eucalyptus Globulus Labill (Tasmanian bluegum) Plantations In Southwest Western Australia

Honi M. Adolphson
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**ACARI (MITE) ABUNDANCE AND DIVERSITY IN SOIL AND
LITTER LAYERS OF *EUCALYPTUS GLOBULUS* LABILL.
(TASMANIAN BLUEGUM) PLANTATIONS IN SOUTHWEST
WESTERN AUSTRALIA**

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Thesis submitted in partial fulfilment of the requirements for the award of
Master of Science (Biological Science)

**School of Natural Sciences
Edith Cowan University**

July 2000

“Ecologists have the responsibility to provide society with the knowledge essential for both understanding and wisely managing the earth and its biological resources”

David Tilman, 1999

ABSTRACT

With the increased planting of *E. globulus* monoculture plantations, concerns surround the impact these mass plantings will have on the soil environment and wider ecosystem. Soil and litter mites are the dominant saprophages of terrestrial ecosystems, contributing to decomposition processes through their comminution and grazing activities. Despite the importance of mites to decomposition processes, there have been no investigations to date of the litter and soil communities under these plantations within Australia.

This study investigated the impact of *Eucalyptus globulus* subsp. *globulus* (Tasmanian bluegum) monoculture plantations on the diversity and abundance of the soil and litter acarine (mite) fauna. Mite communities under three 8 year-old *E. globulus* plantations sited on reclaimed pastureland were compared with an adjacent native *E. marginata* (jarrah) forest and a grazed pasture in the mediterranean-type region of southwest Western Australia. Sites were sampled in spring 1997 and new sites randomly selected in autumn 1998.

Large seasonal variations in abundance and diversity were found between the sampling periods of spring 1997 and autumn 1998, influenced considerably by soil moisture. Species richness was consistently higher in both the soil and litter layers under native jarrah forest, with the *E. globulus* plantations intermediate in species richness between the native forest and grazed pasture. A total of 114 mite species/morphospecies was recognized; 16 Mesostigmata, 52 Prostigmata, 45 Oribatida (Cryptostigmata) and 1 Astigmata. Species diversity in the surface litter was higher under the native forest, reflecting the greater

heterogeneity of the litter. Diversity in the underlying soil was similar for the native forest and *E. globulus* plantation, although there were considerable differences in species richness. Soil diversity in the pasture was very poor in comparison to the two forest systems, reflecting the absence of a litter layer and reduced niche dimensions.

The monospecific *E. globulus* litter possibly reduced niche variety by simplifying the physical habitat and reducing the variety of resources available for the saprophagous fauna, of which the oribatid mites are a major component. Consequently, the number of families represented in the microphytophagous, phytophagous and predator functional groups under the *E. globulus* plantation may have been reduced. The most common oribatid families under the plantation were those often reported from disturbed habitats, Oppiidae and Tectocepheidae, and those common in many habitats, Oppiinae and Oppiellinae.

An important management implication arising from this study is that these plantations would harbour a more diverse mite fauna if planted as a mixed tree stratum.

USE OF THESIS

The Use of Thesis statement is not included in this version of the thesis.

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CHAPTER 1: INTRODUCTION

1.1 Eucalypts in Plantation Forestry

1.1.1 The Australian Eucalypt

The genus *Eucalyptus* belongs to the Myrtaceae family, of which about 500-600 currently recognizable species are endemic to Australia (Millet, 1969; Pryor, 1976). Occurring within all areas but the driest parts of the Australian continent, the genus has a remarkable environmental and vegetational range as a result of selection pressures and evolutionary responses endured by the continent (Florence, 1996). Eucalypts dominate the Australian sclerophyll forests that extend in a continuous coastal band from the Tropic of Capricorn to South Australia and Tasmania, with a second area in the southwestern part of Western Australia (Ovington, 1983).

The eucalypt gene resource in Australia is of world importance because of the potential capacity of eucalypts to significantly increase global commercial timber reserves (Ovington and Pryor, 1983). Eucalypt species form some of the most productive stands in exotic environments; e.g. maximum rates of dry matter production can exceed $40 \text{ t ha}^{-1}\text{yr}^{-1}$ (Jarvis and Leverenz, 1983, cited in Beadle and Turnbull, 1992). Their adaptability, extremely high potential growth, resprouting ability, and the wide utility of the wood makes eucalypts highly desirable for mass growth (Beadle and Turnbull, 1992; Mitchell, Ford-Robertson, Hinckley, and Sennerby-Forsse, 1992).

1.1.2 Eucalypt Plantations: The World View

Plantations can be used as either supplements or alternatives to native forests (Mitchell *et al.*, 1992; Stephens, Hansard and Dean, 1993). The development of fast-growing eucalypt plantation forests is important for sustainable management and efficient use of native forests. Eucalypts are one of the most important trees planted in the middle and lower latitudes of the world, those commonly considered as the mediterranean climate regions. These regions are characterized by dry, hot summers and wet, mild winters with a more or less well defined summer drought (di Castri, 1981). Annual rainfall ranges from 250 to 1200 mm per year (Hobbs, 1992). The five widely disjunct mediterranean regions of the world are the Mediterranean Basin, California, central Chile, the southern and south-western Cape Province of South Africa, and south-western and parts of southern Australia (di Castri, 1981; Hobbs, Richardson and Davis, 1995b). Di Castri (1981) gives a more detailed account of these regions and their classification based on the length of the summer drought.

Only a few eucalypt species from this vast genus form the majority of the monoculture plantation estate both within and outside Australia (Beadle and Turnbull, 1992). The most prominent are *Eucalyptus camaldulensis*, *E. citriodora*, *E. delegatensis*, *E. globulus*, *E. grandis*, *E. robusta*, *E. saligna*, *E. tereticornis*, *E. urophylla* and *E. viminalis* (Florence, 1996). Following the classification of Pryor and Johnson (1971), all these species belong to the subgenus *Symphyomyrtus*. This is the largest subgenus, with approximately 370 species (Beadle and Turnbull, 1992).

Eucalypt plantations are now established in Argentina (Dalla-Tea and Marco, 1996), Australia, Brazil (Félix de Barros and Ferreira de Novais, 1996), California, Chile (Prado, Toro and Jorge, 1996), China (Huoran and Wenlong, 1996), Ethiopia (Pohjonen and Pukkala, 1990), India (Negi and Sharma, 1996), Uruguay, Peru, Bolivia, Ecuador (Turnbull and Pryor, 1984), New Zealand (Attiwill and Adams, 1996), Portugal (Pereira, Tomé, Madeira, Oliveira, Tomé and Almeida, 1996), and South Africa (Ledig, 1989; Herbert, 1996).

1.1.3 Plantation Forestry in Australia

Native eucalypt forests are a vital component of the Australian landscape and are important for the conservation of wildlife, the protection of water catchments, prevention of soil erosion and species and ecosystem biodiversity (Ovington and Pryor, 1983). Yet, despite their importance, these forests continue to be utilized as the backbone of Australia's timber industry, supporting the domestic demand for sawn timber and pulpwood and an overseas demand for woodchips (Ovington and Pryor, 1983; Beadle and Turnbull, 1992). The significant decline of available native eucalypt forests in Australia has increased the need to establish both softwood and hardwood plantations (Stephens, *et al.*, 1993). By March 1991, ninety per cent of the plantation forest area in Australia consisted of softwood (coniferous) plantations, while the remaining ten per cent was hardwood (Stephens, *et al.*, 1993).

Plantations of exotic coniferous tree species, principally of the genus *Pinus*, were established in Australia late in the 19th century to compensate for the inadequate supplies of softwood timber from native species (Cromer, 1996). Approximately 0.5 million ha have been planted on some of the "least-desired" soils of the continent, soils which carried poor-

quality eucalypt forest and those that would have required special fertilizer treatment if converted to agriculture (Attiwill and Leeper, 1987).

The development of hardwood plantations in Australia dates back to the 1950's, when several experimental eucalypt plantations were established as supplements to native hardwood supplies (Cromer, 1996). Since then, the establishment of fast-growing plantation forests, also known as short-rotation forestry, has occurred relatively quickly. At the end of 1993, the eucalypt plantation estate was approximately 125 000 ha, with new plantations being established at a rate of about 15 000 ha per annum and increasing (Cromer, 1996). The largest areas of plantings are in Tasmania (38% of the total area of plantations within Australia occur here), Western Australia (20%) and Victoria (18%) (Cromer, 1996). The eucalypt plantation estate is primarily *Eucalyptus globulus* Labill..

1.2 *Eucalyptus globulus* Labillardiere

Four subspecies exist within the *E. globulus* Labill. complex. These are *E. globulus* Labill. subsp. *bicostata* (Maid *et al.*) Kirk. (common name Southern bluegum), *E. globulus* Labill. subsp. *maidenii* (F.Muell) Kirk. (common name Maidens gum), *E. globulus* Labill. subsp. *psuedoglobulus* (Naudin ex Maid) Kirk., and *E. globulus* Labill. subsp. *globulus* (common name Tasmanian bluegum) (Anon., 1979). Each subspecies is important in forestry, however, *E. globulus* subsp. *globulus* will be the focus of this review and will be written as *E. globulus* hereafter.

1.2.1 *Eucalyptus globulus*: The Tasmanian bluegum

Eucalyptus globulus Labill. is a tree of worldwide economic importance. The species was the first eucalypt to become widely known outside Australia (Anon., 1979), with exotic plantings first recorded in Europe in the early 1800's (Millet, 1969). Early successes, mainly because the juvenile leaves are not palatable to livestock (Anon., 1979), have led to *E. globulus* being the most extensively planted temperate eucalypt species in the world, specifically in the mediterranean-type regions (Cromer, 1996). The expansion of plantings worldwide has increased the range of site conditions encountered by the species, demonstrating wide ecological tolerance (Bennett, Weston, and Attiwill, 1997). The natural occurrence of *E. globulus* is within Australia, predominantly on the southern and eastern sides of Tasmania. The species is also found on Flinders and King Islands in Bass Strait, and on Wilson's Promontory, Cape Otway and in the Strzelecki Ranges in southern Victoria (Kirkpatrick, 1975).

E. globulus is restricted to a range of habitats within a particular area (Kirkpatrick, 1975). Growing in coastal or hilly localities, *E. globulus* prefers moderate summer temperatures (without hot, dry winds) and a high winter rainfall pattern of 500-1500 mm per annum (Millet, 1969). Moister gullies of the warmer parts of the range are favoured; from sea level to 1,000 feet in Tasmania and to about 1,500 feet in Victoria (Millet, 1969; Kirkpatrick, 1975). *E. globulus* thrives best on heavy soil or good quality loam with adequate moisture. The species exhibits a local distribution pattern that is seldom found in extensive pure stands, usually being associated with *E. obliqua*, *E. rubida*, and *E. regnans*

in favourable conditions. On poorer soils the species occurs with *E. viminalis*, *E. linearis*, *E. risdoni*, and *E. amygdalina* (Millett, 1969).

Tree height in native regions reaches 45-55 metres by a straight, massive trunk and a heavy, spreading crown (Anon., 1979). Being an evergreen plant, leaf litterfall occurs continuously throughout the year. Juvenile leaves are large, blue-silver in colour and are retained for 3-4 years, whilst mature leaves are long and sickle-shaped. The bark is rough, grey, and persistent at the base, and smooth above. The timber is a characteristic light yellow-brown colour and is of open texture, often with an interlocked, spiral grain. It is also strong, heavy and moderately durable, yet difficult to season and work (Millett, 1969; Anon., 1979).

The fast growth of *E. globulus* throughout the early years and its intense productivity response to increased nutrients (Pereira, 1992) lends the species to short-rotation silviculture. Trees are harvested on a rotation of around 8-12 years (Miller, 1989), after which they are vigorously coppiced for a second harvest (Eckersley, Ellis and George, 1993).

The major uses for *E. globulus* plantations are as wood fibre for the manufacture of pulp and the production of high quality paper (Cromer, 1996). In developing countries *E. globulus* is exclusively grown for fuelwood and solid wood products (Pohjonen and Pukkala, 1990; Daoqun, Huoran, and Yongqi, 1992).

1.3 *Eucalyptus globulus* in Western Australia

1.3.1 Southwest Western Australia: Climate and Biota

The south-western part of Western Australia is one of the worlds' five mediterranean regions (di Castri, 1981). The region consists of both a subhumid (5-6 month summer drought) and a humid (3-4 month summer drought) mediterranean climate correlating with the sequence of anticyclonic and cyclonic air masses that pass across southern Australia (di Castri, 1981). The region, along with those of southern Australia and South Africa, differ from the other mediterranean regions because summer rainfall is common, and the very old soils are weathered and relatively poor in nutrients (di Castri, 1981; Hobbs, Groves, Hopper, Lambeck, Lamont, Lavorel, Main, Majer and Saunders, 1995a).

The mediterranean region of southwest Australia shows great heterogeneity in plant and animal communities, landforms and soil types (Hobbs *et al.*, 1995b). The majority of the region is underlain with a mantle of infertile, eroded, lateritic soil over a granite basement (Florence, 1996). Ecosystem types range from closed forests to open savannah woodlands and shrublands (Beard, 1983, as cited in Hobbs, 1992), with a large part of the region being covered by dry sclerophyll forest, specifically *E. marginata* and *E. diversicolor* (Florence, 1996). Plant species diversity and levels of endemism are both high. Large numbers of plant species have restricted ranges resulting in the region containing the highest numbers of rare, endangered and restricted plant species in Australia (Hobbs, 1992).

Southwest Western Australia has undergone rapid transformation due to land clearance, forestry, mining and the introduction of exotic organisms (Hobbs, 1992). In particular, the transformation of the natural vegetation to predominantly agricultural land occurred rapidly compared with the Mediterranean Basin (Hobbs, 1992). As a consequence, we are left with a highly fragmented landscape of many small remnants of native vegetation embedded in a predominantly agricultural matrix (Hobbs, 1992; Hobbs *et al.*, 1995a).

1.3.2 History of *Eucalyptus globulus* in Southwest Western Australia

Up to the early 1980's, the only tree species used specifically for wood production in Western Australia was *E. diversicolor* F. Muell. (karri). Over the past 10 years, the preference has been to plant *E. globulus* Labill. on cleared farm-sites. It is now the major species planted in the state. At the end of 1995 an estimated 40,000 hectares had been planted (Ritson, 1995), with the rate of new planting expected to be at least 5000 ha yr⁻¹ (Cromer, 1996). At the same time, the production of pulpwood from native forests is expected to decline over the next 5-10 years. The current aim is to increase the range of sites that can be planted with *E. globulus* using selective breeding (Cromer, 1996).

Both private industry (Bunnings Treefarms Pty Ltd) and the state government (Department of Conservation and Land Management) promote the establishment of *E. globulus* plantations in Western Australia as a means to supplement and eventually replace the export woodchip supply from native forests (Cromer, 1996). In addition, pulpwood plantations are being promoted as multiple-purpose timber-belts to provide land-care benefits and wood production, whilst improving soil sustainability by moderating local

micro-climates, lowering the water tables, and reducing soil and streamwater salinity (Cromer, 1996). Plantations are now established on ex-pasture sites, specifically placed to make the greatest contribution to productivity and sustainability of agriculture in this way. Although not native to Western Australia, *E. globulus* is a desirable plantation species as it grows rapidly on a range of sites, has versatile wood, coppices vigorously and is resistant to the major pathogen of jarrah, the dieback fungus *Phytophthora cinnamomi* (Turnbull and Pryor, 1984; Cromer, 1996).

Like other monocultures, *E. globulus* plantations are major ecological disturbances likely to affect the biodiversity and abundance of native fauna. It is only in the past two decades that any attention has been focused on the effect of these landscape modifications on the soil-litter stratum. Most investigations to date have concentrated on the management regimes of dryland agriculture, especially comparisons of conventional and non-tillage systems (Anderson, Knight and Elliott, 1991). Very little emphasis has been placed on the effects that monoculture tree species have on the soil mesofauna and the wider soil environment. This should be of concern, considering both the direct and indirect roles the soil biota has in regulating primary production.

1.4 The Soil Environment

1.4.1 Importance for Ecosystem Function

The soil is an important component of the earth's biogeochemical processes, influencing the functions and productivity of the immediate terrestrial ecosystem and the wider

environment (Doran and Parkin, 1994; Beare, Coleman, Crossley, Hendrix and Odum, 1995). Soil is a complex ecological system, playing roles in the production of food and fibre (Ahern and Yen, 1977; Doran and Parkin, 1994). In an ecosystem context, the capacity of soil to function and sustain biological productivity, maintain environmental quality, and promote biotic functions, relates specifically to its ability to function as a medium for plant growth, to partition and regulate water and nutrient flows in the environment, and to act as an environmental buffer (Doran and Parkin, 1994; Larson and Pierce, 1994).

Up to 90 per cent of aboveground biomass falls directly to the soil surface (Coleman and Crossley, 1996). Plants rely on soil biological processes to replenish the nutrients lost through the production of plant litter. These processes, such as decomposition and nutrient cycling, are crucial to continued productivity of terrestrial ecosystems (Seastedt, 1984). The highly connective nature of foodwebs implies that most, if not all, terrestrial organisms depend either directly or indirectly on these processes (Stork and Eggleton, 1992).

One of the major variables affecting soil processes is organic matter, which is critical to the regulation and functioning of the soil-litter stratum. Organic matter largely determines the patterns of distribution of soil animals, their aggregation, the relative abundance of dominant groups and the values of species diversity, as well as the physical properties of the soil habitat (di Castri, 1973; Spain and Hutson, 1983). This is particularly true for mediterranean vegetation where the production of organic matter is low (di Castri, 1973). Organic matter characteristics depend ultimately on the activities of soil organisms. Organic

residues are decomposed and incorporated in the soil by the combined activities of microorganisms and soil fauna (di Castri, 1973; Richards, 1974).

Decomposition is broadly expressed as a progressive decrease in the mass of a substrate, resulting in a change of state under the influence of biological and abiotic factors (Seastedt, 1984; Swift, Heal and Anderson, 1979). It is important in the reduction of organic residues to inorganic matter (mineralisation), the release and recycling of nutrient elements, and the elimination of detritus build-up (Reichle, 1977; Swift *et al.*, 1979). The process impels complex belowground food webs (Coleman & Crossley, 1996). In a geological time scale, decomposition serves to return elements to the biogeochemical cycles of the biosphere (Reichle, 1977; Coleman and Crossley, 1996).

Decomposition can be measured as a decrease in the mass of organic substrate. This decrease is caused by (1) leaching of soluble materials of the intracellular components, (2) the physical breakdown of the detritus, known as comminution, and (3) catabolism or oxidation of the organic matter into fixed carbon dioxide, water and energy (Swift *et al.*, 1979; Seastedt, 1984). The decomposition of organic matter and the consequent mineralisation of its nutrients is the essence of nutrient cycling (Attiwill and Leeper, 1987), and is significantly dependant on the quantity of plant detritus available for decomposition and the decomposition rate (Lee, 1991). Interactions between the abiotic environment, soil fauna, microorganisms and plants control patterns and rates of organic matter decomposition, nutrient cycling, nutrient immobilization and mobilization, all of which are paramount to the regulation of soil function and productivity (Lee, 1991).

1.4.2 Biotic Components of the Soil and Litter

Microflora

Bacteria and fungi constitute the microflora and are the primary decomposers of organic matter. Both use exoenzymes to decompose insoluble substrates; hence their activity is central to the circulation of inorganic nutrients. The small size of the microflora denotes a very large surface-to-volume ratio that allows rapid interchange of materials between their cells and the environment and thus extremely rapid colonization of organic matter (Richards, 1974).

Fungi are able to breakdown cellulose plant remains, which constitute a major fraction of the organic matter. Mechanical pressure from their elongating hyphae permits exploration of dead plant tissues with relative ease and supplements the action of exoenzymes (Richards, 1974). Fungi also play important roles in soil fertility and primary production as root pathogens and as participants in symbiotic associations with plant roots (Klironomos and Kendrick, 1995). Bacteria contribute to decomposition via oxidation of organic compounds. Usually unicellular or in clustered colonies, bacteria occupy discrete patches of soil measuring only a few cubic centimetres in volume (Coleman and Crossley, 1996).

Soil Fauna

The soil fauna are a diverse group of organisms ranging in size from a few micrometres to several centimetres in diameter. Body size provides a useful classification resulting in three major groupings: microfauna, mesofauna, and macrofauna (Swift *et al.*, 1979). It is the size of the soil fauna, their mobility, feeding strategies and population densities that determine

their influence on soil processes (Anderson *et al.*, 1991; Linden *et al.*, 1994). As major components of the decomposer food web, soil animals regulate decomposition and nutrient mineralisation and immobilization processes (Meyer, 1995; Parmelee, 1995).

The microfauna, with sizes less than 200 μm , comprise the protozoa and small nematodes, and appear to have a principle role in soil processes via interactions with the microflora. These organisms inhabit water films or water-filled pore spaces in soils. Protozoa are known to be important bacterivores, fungivores and even nematophages, as nematodes may be important as regulators of bacterial populations (Coleman and Crossley, 1996).

Soil mesofauna, or microarthropods as they are often referred to, are animals 100 μm to 2 mm in length, and consist of mites (Acari), springtails (Collembola), various small insects (Insecta), spiders (Araneida) and pseudoscorpions (Pseudoscorpionida) (Linden *et al.*, 1994). The members of this diverse group of invertebrates are considered to be sufficient in size to overcome the surface tension of water on soil particles, yet are not large enough to disrupt the soil structure by their movements through existing air-filled pore spaces (Stork and Eggleton, 1992; Coleman and Crossley, 1996).

Animals such as earthworms, larger Enchytraeidae, millipedes (Diplopoda), centipedes (Chilopoda), scorpions (Scorpionida) and larger spiders constitute the macrofauna, with sizes measured in centimetres. The relative larger size of these animals affects the overall physical structure of the soil (Coleman and Crossley, 1996). In particular, earthworms play a direct role in decomposition by fragmentation and mixing of plant material through the

soil. In addition they promote microbial activity, their faeces acting as substrates for microbial growth (Richards, 1974).

Soil faunal populations contribute significantly to rates of soil nutrient turnover, soil texture and consistency, the formation of stable soil aggregates, porosity, the infiltration of water, and soil water retention characteristics (Reichle, 1977; Seastedt, 1984; Lee and Foster, 1991). These activities are affected by chemical and physical properties of the soil. The abiotic characteristics in turn influence faunal populations (Lee and Foster, 1991).

The study of soil faunal communities in mediterranean-type regions is based predominantly on the soil mesofauna, particularly the mites and springtails (Wallwork, 1967; Parmelee, 1995), because they represent the large majority of animals collected through the most commonly used Berlese-Tullgren funnel extraction method (di Castri and Vitali-di Castri, 1981). As major components of the soil (Seastedt, 1984), the mesofauna are considered biological indicators of change in many terrestrial ecosystems because of their high abundance, their considerable taxonomic and trophic diversity, and their importance to soil biological processes (Linden *et al.*, 1994; Pankhurst *et al.*, 1995; Parmelee, 1995).

1.5 The Soil Mesofauna, with particular reference to the Soil Acari

Soil mesofauna, principally mites and springtails, dominate the soil fauna in terms of abundance, with populations of each group often reaching 100,000 or more per square metre of soil surface. Biomass is low, being a few g/m² of soil surface (Richards, 1974; Seastedt, 1984; Lee and Foster, 1991; Linden *et al.*, 1994). Soil mesofauna represent all

trophic groups including bacterivores, fungivores, herbivores, omnivores and predators (Parmelee, 1995). The majority, however, are saprovores that feed on decaying and dead plant litter and roots and associated micro-organisms (Lee and Foster, 1991). The mites make up the greater proportion, comprising between 60-90 per cent of the microarthropod fauna.

1.5.1 Classification of Acari

Soil mites are classified into four major groups distinguished by the character of the respiratory system in adults (Krantz, 1978; Wallwork, 1967), the Mesostigmata, Prostigmata, Cryptostigmata (also called Oribatei or Oribatida) and Astigmata.

Mesostigmatid mites are predominantly predaceous, often selecting specific prey, such as nematodes, small enchytraeid worms, insects or other mites. These typically large mites are usually active and free ranging (Richards, 1974), and usually reach peak abundances in moist, rich soils where litter characteristics are relatively stable (Hutson and Veitch, 1983).

Prostigmatid mites are the largest and most variable of the groups. They represent all trophic niches from large microarthropod predators through to small fungivorous and nematophagous species (Krantz, 1978). They too, are free ranging and active (Richards, 1974), being particularly abundant in dryland ecosystems.

The poorly specialized fungivorous and detritivorous Cryptostigmatids are numerically the most abundant group in mor type forest soils (where there is a sharp boundary between the

litter layers and mineral soil), especially in the fermentation layer where active decomposition takes place (Richards, 1974; Coleman and Crossley, 1996). Their distribution, like that of Collembola, is governed by the need for habitats with a saturated atmosphere (Richards, 1974). They are also the most biologically important, as regards deep mixing of organic material (Wallwork, 1967; Seastedt, 1984; Meyer, 1995; Coleman and Crossley, 1996).

The Astigmata are associated with rich, decomposing nitrogen sources. They tend to be somewhat rare in forested soils, yet increase numerically in agricultural and disturbed or modified soil systems (Coleman and Crossley, 1996).

Soil mites are predominantly hemiedaphic, being vertically distributed in the organic layers associated with the top 10 cm of the underlying soil (Lee and Foster, 1991). Their distribution, however, extends into the other two distinct ecological zones, namely the epigeal vegetation zone and the deeper mineral strata of the soil (Wallwork, 1967).

1.5.2 The Collembola

Soil Collembola (springtails) and oribatid mites, in addition to their dominance in the soil, occupy similar ecological niches. It is therefore difficult to separate one group from the other as regards their role in the soil. Springtails occur throughout the soil profile where they have varied feeding habits. Springtails may devour bacteria, fungal hyphae and spores, decomposing organic matter, faeces, living plants or animals (Richards, 1974). They are

active in fragmenting litter, enhancing colonization by fungi and other microbes (Richards, 1974; Coleman and Crossley, 1996).

1.5.3 The Roles of Mesofauna in the Soil-litter Stratum

Mites and springtails play indirect, yet significant roles in soil processes. Their effects include fragmentation of litter and conversion to faeces, mixing of litter and soil and regulation of microflora and microfaunal abundance and diversity (Crossley, 1977).

One of the most important effects of mesofaunal feeding activities is comminution, the reduction in particle size of organic matter (Swift *et al.*, 1979). This physical breakdown of litter is important in exposing greater surface areas to both microbial colonization and activity and leaching (Seastedt, 1984; Majer, 1987; Linden *et al.*, 1994).

The vertical movements of mites assist decomposition by transporting organic matter, microbial propagules and faecal pellets and cast products to the lower layers and root zones of the soil profile. This in turn promotes the growth of fungal and bacterial populations throughout the soil profile (Wallwork, 1967; Parmelee, 1995). The dissemination of fungal spores by fungivorous mites, either by readily adhering to the body surface or by transportation in the gut, assists decomposition by promoting microbial growth and distribution (Wallwork, 1967; Parmelee, 1995). Faecal pellets, generally less than 1 mm in diameter and often very numerous in the soil, are frequently associated with large colonies of bacteria lying in the adjacent soil, probably stimulated to fresh growth and division by soluble metabolites leaching from the pellet (Lee and Foster, 1991). Alteration of the

microbial microenvironment significantly affects the formation of soil aggregates, pore space, and soil stability (Lee and Foster, 1991; Parmelee, 1995).

Mesofaunal feeding on fungi and bacteria can directly affect nematode populations. Soil nematodes participate indirectly in decomposition by regulating the microbial populations whilst they in turn are a food source for other biota (Richards, 1974). Fluctuations in microbial numbers directly affect nematode populations.

1.5.4 Acari as Biological Indicators

Consideration of the role of mites in ecosystem processes and their interactions with other soil biota (Crossley *et al.*, 1992) is increasing, along with the assessment of mites as biological indicators of ecosystem perturbation and recovery (Meyer, Campbell, Moser, Hess, Rawlings, Peck and Heck, 1990; Paoletti and Bressan, 1996; van Straalen, 1998). Changes in mite activity may be indicative of changes in soil health and quality (Stork and Eggleton, 1992; Pankhurst *et al.*, 1995; Paoletti and Bressan, 1996).

Oribatid mites, in particular, offer several advantages for assessing the quality of terrestrial ecosystems (Behan-Pelletier, 1999). Oribatids have both high density and species diversity (about 10,000 species have been described worldwide; Norton, 1990; Colloff and Halliday, 1998), and because of their various levels of environmental sensitivity, are excellent for assessing ecosystem perturbation and recovery (Aoki, 1979; Hunt, 1994). Living in the organic horizons, oribatids are easily sampled and can be sampled in all seasons. Oribatid mites also represent a trophically heterogeneous group. Few oribatids have modifications

for dispersal, and are therefore unable to easily escape from stressful soil conditions (Norton, 1994). Some oribatid family level taxa appear to have more potential in biodiversity studies due to their high abundances, are readily recognizable and /or are better known at the species level (Hunt, 1994). Oppiidae, Oribatulidae, Galumnidae and Brachychthoniidae (among others) may be useful in biodiversity assessments and as bioindicators due to their particular sensitivities (Hunt, 1994). However, Aoki (1979) rated some oribatid families (Oppiidae, Oribatulidae, Haplozetidae and Galumnidae) as insensitive to environmental conditions.

1.6 Drawbacks in working with Acari

Several inherent difficulties are encountered when working with mites. Mites are too commonly ignored or lumped into higher taxa groupings (Crossley *et al.*, 1992) because of the difficulties arising from their taxonomic and microscopic nature. Since the most effective indicators are species (Behan-Pelletier, 1999), the incomplete taxonomy of mites is a limitation. There are over 40,000 named species of Acari worldwide, with fewer than 3000 mite species in 240 families (Astigmata 330, Oribatida 330, Prostigmata 1270, Mesostigmata 675) described for Australia, accounting for only 3 % of the total acarofauna (Halliday, 1998; Walter and Proctor, 1999). The Australian fauna is well described for only a limited number of mite groups. Almost one-third of the total described mite fauna of Australia occurs in only 10 families (mostly Mesostigmata and Prostigmata) including a total of 894 species (Halliday, 1998). The dominance of these families arises because of both economic significance and their parasitic associations with vertebrates, invertebrates and plants. Little is known of the biology and ecology of free-living mites due to the

difficulty in studying trophic status and feeding niche as selectivity of food material may vary seasonally, with habitat, with microhabitat and with ontogenetic shifts as the animal ages (Giller, 1984). The very small, cryptic nature of these soil creatures also makes the construction of food webs that reflect actual trophic behaviour very difficult (Walter, Hunt and Elliot, 1988; Stork and Eggleton, 1992). Finally, the time required to process soil and litter mesofauna is exceptionally lengthy (Lawton *et al.*, 1998; Bisevac and Majer, 1999). Lawton *et al.* (1998) made inventories of the number of 'scientist-hours' required to process ecological samples including fieldwork, extraction, sorting to morphospecies, identification where possible and data compilation. The proportion of morphospecies that cannot be assigned to named species and the number of 'scientist-hours' both increase for smaller-bodied taxa, e.g. the time required to process termites and soil nematodes approximate 2,000 and 6,000 hours, respectively (Lawton *et al.*, 1998).

1.7 Effects of Disturbance on Acari

The effect of agricultural operations, such as cultivation, tillage, rotation, fertilization, mowing and irrigation, on soil invertebrate populations has been commonly studied (Coleman, 1985; Siepel and Van De Bund, 1988; Stinner and House, 1989; Mueller, Beare and Crossley, 1990), yet is limited in Australian agricultural systems (Pankhurst *et al.*, 1995). Mesofaunal population densities and species diversity have been observed to be lower under conventional ploughing practices than those with no-tillage treatments (Stinner, McCartney and van Doren, 1988; Crossley *et al.*, 1992; Gupta, 1994). Pankhurst *et al.* (1995) describe a series of soil biological properties that they evaluated as potential bioindicators of soil health for cropping systems in southern Australia. A variety of soil

organism populations, including mites and springtails and soil processes mediated by soil organisms, were examined. Populations of both mites and springtails were affected by the management practices evaluated, in particular the tillage-stubble management significantly affecting springtails (among others). Oribatid heterogeneity, at the species level, was significantly reduced in the total conventional plow-based fields and reduced-till fields as compared to the no-till practices examined by Franchini and Rockett (1996). A general decrease in both abundance and diversity may occur when soil is plowed and planted with crops (cited in Franchini and Rockett, 1996). Other agroecosystem practices such as compaction of soils by tractors also reduces invertebrate biomass and abundance (Aitajet *et al.*, 1977).

The effect of sheep stocking intensity on the soil mesofauna in southeastern Australia has been studied at length (King and Hutchinson, 1976 and 1983; Hutchinson and King, 1980). King and Hutchinson (1983) found the accumulative effects of sheep grazing over time decreased soil mite abundances. In a 1985 study (King, Greenslade and Hutchinson) increased sheep numbers (hence increased grazing intensity) decreased springtail proportions and species richness. Kay, Sobhy and Whitford (1999) studied microarthropod communities along livestock grazing disturbance gradients, inside and outside grazing enclosures and on areas subjected to restoration efforts (herbicide and bulldozing) in the Chihuahuan Desert rangelands. Results showed mite numbers generally increased with decreasing grazing disturbance.

The use of soil invertebrates as bioindicators of soil contamination has also been widely assessed (Paoletti *et al.*, 1991). Springtails, oribatid mites and other Acari have been

examined for heavy metal content (Bengtsson and Gunnarsson, 1984; Janssen, 1988; Tranvik *et al.*, 1993). Wade *et al.* (1980) demonstrated that soil invertebrates react to pollution near roadsides and the effect of pollution of urban soils by traffic exhausts has reduced total oribatid numbers and species richness (Stamou and Argyropoulou, 1995). The effects of pesticides on soil mesofauna have also been assessed (Thompson and Edwards, 1974; Eijsackers and van de Bund, 1980).

Soil invertebrates have also been used for monitoring the effects of logging and fire in Australian forests. Springett (1976) found both the number of mesofaunal species and population densities were reduced after burning in native jarrah and karri forests in Western Australia. Majer (1984) investigated the short-term responses of soil and litter invertebrates to a cool autumn burn in jarrah forest. Litter mites were reduced by fire after a long time lag and the soil springtails were initially stimulated but later declined following fire. Neumann (1991) studied the effects on epigeal (above-ground) arthropods of high-intensity wildfire plus salvage logging, and of harvesting by clearfelling plus slash burning in commercial Mountain Ash (*E. regnans*) regrowth forest in southeastern Australia. The replacement of the ecologically complex regrowth forest by much simpler even-aged regeneration caused instability among the litter-frequenting arthropods, with an immediate decline in species diversity. In addition, Bird and Chatarpaul (1986) found whole-tree harvesting (the removal of all aboveground biomass of trees including branches and leaves) of forestry systems had more effect on soil arthropods than conventional tree harvesting.

1.8 Impact of Forest Monocultures on Soil Fauna, with particular emphasis on *Eucalyptus globulus*

The extensive removal of native vegetation and the conversion from indigenous to introduced vegetation has profound effects on many ecosystem components and processes (Hobbs *et al.*, 1995a). Consequent reduction in landscape complexity and heterogeneity has implications for the conservation of species diversity, particularly with regards to plant-animal-soil inter-relationships where one or several species are always dependent on another (Ovington and Pryor, 1983). Reduced biodiversity is the result of simplification of ecosystem structure and function (Hobbs *et al.*, 1995a). Comparative studies that attempt to link diversity and ecosystem processes provide valuable information on the likely significance of biodiversity. There are often clear decreases in soil faunal diversity along the gradient of natural to the modified (grazed natural lands) to replaced (exotic plantations and pastures etc. in systems cleared of native vegetation) environments (Springett, 1976b; Ahern and Yen, 1977; di Castri and Vitali-di Castri, 1981; Greenslade and Greenslade, 1983; Serrelheiro and Madeira, 1990; Ananthakrishnan, Gopihandran and Subramaniam, 1993).

Populations and community structure of soil biota are strongly affected by changes in vegetation (di Castri, 1973), in which vegetation replacement causes a redistribution of organic matter within the soil profile and the buildup of substantial litter layers. Modified environments, such as monoculture plantations, are known to acquire an impoverished litter fauna (Springett, 1976b; Spain and Hutson, 1983). Springett (1976b) compared the abilities of soil biota to decompose leaf litter and cellulose fibre under *Pinus pinaster* plantations

and indigenous woodland in Western Australia. Reduced species diversity and an impoverished soil and litter fauna characterized the pine plantations, resulting in a reduced ability to decompose leaf litter leading to a reduced rate of energy flow and of nutrient cycling.

Investigations of soil and litter fauna in *E. globulus* plantations are limited, despite the mediterranean regions boasting some extensive studies of these monocultures. Current areas of research deal mainly with the relationship between tree growth and nutrient and water availability, with the emphasis being on improving growth under plantation conditions for economic reasons (Pereira, 1992; Cromer, 1996). Investigations concerning nutrient/fertilizer applications and irrigation regimes are foremost (Cromer and Williams, 1982; Schönau and Herbert, 1989; Madeira, Araújo, and Pereira, 1995; Judd, Bennett, Weston, Attiwill, and Whiteman, 1996) for optimization of biomass production (Cromer and Williams, 1982; Fabiáo, Persson and Steen, 1985; Pereira, Araújo, Borralho and Leal, 1987; Pereira, Linder, Araújo, Pereira, Ericsson, Borralho, and Leal, 1989; Inions, 1992; Fabiáo, Madeira, Steen, Katterer, Ribeiro, and Araújo, 1995; Kätterer, Fabiáo, Madeira, Ribeiro, and Steen, 1995). Nutrient uptake, litterfall, nutrient release and cycling within *E. globulus* plantations are also well studied (Upadhyay, 1982; Pereira, Miranda and Melo, 1987; Miller, 1989; George and Varghese, 1990; Lisanework and Michelson, 1994). The effects of *E. globulus* plantations on soil fertility in some mediterranean regions (Michelson, Lisanework, Friis, and Holst, 1996), as well as changes in soil properties under these trees (Madeira, 1989; Madeira and Pereira, 1990/91) have been studied because they have been blamed for a decrease in soil nutrients. Such outcomes arguably influence the soil and litter fauna either directly or indirectly.

Studies with the emphasis on the mesofauna under these plantations are few. Di Castri and Vitali-di Castri (1981) noted mean densities of the main soil arthropod groups extracted from natural and human-modified ecosystems in Chile, including a sclerophyllous forest, *E. globulus* and *Pinus radiata* plantations, and irrigated pasture. Springtail abundances were higher and mite abundances were lower under the *E. globulus* plantation in comparison to the native forests. Species diversity was relatively low under *E. globulus*. Serralheiro and Madeira (1990) compared densities of arthropod populations under an *E. globulus* plantation with those under native vegetation dominated by *Quercus suber* in central Portugal and found significantly lower densities in the forest floor (litter) under the *E. globulus* plantation. Species diversity was not measured. Several studies have shown that Collembola populations in Portuguese native forests (*Pinus pinaster* and *Quercus* forests) were more diverse and had higher species richness than those in introduced *E. globulus* plantations (Gama, Vasconcelos and Sousa, 1994; Sousa and da Gama, 1994). Sousa, Vingada, Barrocas and da Gama (1997) studied the impact of *E. globulus* on the Collembola communities of two *Quercus* stands in Portugal, where the substitution of Oak species by *E. globulus* reduced the number of individuals, number of taxa and diversity of Collembola. The study of Pinto, Sousa, Graca and da Gama (1997) also aimed to evaluate if replacement tree species (*E. globulus* and *Acacia dealbata*) would affect native soil collembolan populations.

A review of the literature reveals no investigations of litter and soil faunal communities of monoculture *Eucalyptus* plantations within Australia. The diversity and ecology of ectomycorrhizal fungi under *E. globulus* plantations in Western Australia is an area of

increasing interest (Burgess and Malajczuk, 1989; Grove, Malajczuk, Burgess, Thompson and Hardy, 1991; Burgess, Malajczuk and Grove, 1993; Thompson, Grove, Malajczuk and Hardy, 1994; Lu, Malajczuk, Brundrett and Dell, 1999). Several workers have explored faunal populations under monoculture coniferous forests. Springett (1976b) compared soil microarthropod diversity and abundance and litter decomposition in *Pinus pinaster* plantations with indigenous woodland in Western Australia. The reduced species diversity of litter and soil invertebrates in the pine plantations was associated with a reduced rate of litter decomposition. Ahern and Yen (1977) sampled the soil and litter in *P. radiata* plantations adjacent to native tall open forest in Victoria, and as in Springett's study above, species diversity was reduced in the plantation. Mesofaunal populations in various indigenous Australian forests have been studied (Ahern and Yen, 1977; Hutson and Veitch, 1983; Greenslade, 1985) including several in Western Australia (Springett, 1976a; Koch and Majer, 1980; Majer and Abbott, 1989; Postle, 1989; Cuccovia, 1997). In only two of these studies have the mite fauna been investigated at the species level.

Despite the obvious importance of the saprophagous fauna (of which the mites are the most abundant element) in maintaining ecosystem productivity and the increasing focus on soil mesofauna in *E. globulus* plantations in other mediterranean regions of the world (Di Castri and Vitali-di Castri, 1981; Sousa and Gama, 1994; Pinto *et al.*, 1997; Sousa *et al.*, 1997), no study of the mesofaunal populations have been carried out in Western Australia.

1.9 The Niche Concept: A Useful Theoretical Framework

Attempts to define the niche of a species are difficult and unclear, due in part to different investigators meaning different things by “niche”, yet most ecologists agree that the niche is a central concept of ecology (Real and Levin, 1991) no matter what its level of ecological organization (Leibold, 1995). Key investigators such as Grinnell (1917), Elton (1927) and Hutchinson (1944) have paved the way for modern concepts of the niche.

The overall concept of the ‘ecological niche’ is depicted as the responses of organisms to environmental conditions (Giller, 1984; Begon, Harper and Townsend, 1996; Schoener, 1989). ‘Niche’ describes the way a species population fits into a given community and is part of the whole set of requirements and relationships of the species to the environment (Whittaker and Levin, 1975; Begon *et al.*, 1996). These requirements include those environmental conditions that are necessary for an organism to maintain a viable population and the amounts of each resource that it requires to do so (Begon *et al.*, 1996).

Hutchinson (1957) distinguished two types of niche. The *fundamental* niche involves the overall potential of a species. Morphological limitations and physiological constraints will determine the outer limits to an organism’s occupied position in resource space (Putman, 1994). The *realized* niche includes the more limited spectrum of biotic conditions and resources, which allows a species to maintain viable populations even in the presence of competitors and predators.

Faunal species subdivide resources in three ways. They differ in what they eat (trophic), where they forage (spatial) and when they are active (temporal) (Pianka, 1975). The interaction of resource productivity and the intensity of its use determine the amount of resources available to an organism. Resource dimensions of a species' niche have both upper and lower limits within which a species can thrive (Begon *et al.*, 1996). Broader niches would favour high availability due to high productivity and low use, or low availability due to low productivity or high use (Hespenheide, 1975).

Habitats of species, the kind or range of environments in which a particular species lives, provide numerous niches. Habitats are distinguished by intercommunity variables, such as topographic position, elevation, soil moisture and fertility, type of rock or soil, and so on (Whittaker and Levin, 1975). These niche variables are intensive. They change within a given community and at a particular place, and may change with the type and intensity of biological interactions within the community (Whittaker and Levin, 1975).

The ecological differences in each of the resource dimensions and the ways species relate to other species in the same community should reduce competition and facilitate coexistence (Pianka, 1975; Whittaker and Levin, 1975; Schoener, 1989). The more abundant the resource, the less likely it is that competition will result from its common use (Giller, 1984). Yet if competition is to occur within species due to a lack of available resources, it is more easily resolved by behavioural changes or partitioning of space than by division of the food resource (Hespenheide, 1975). Due to the ease of dividing space, one would expect morphological variation in the population in order to use the different types of habitats.

Thus, differences in dietary variability are likely to be reflected in taxonomic characters related to feeding (Hespenheide, 1975).

1.9.1 Species Diversity and the Niche Concept

Species differ in their occupied niche (Hutchinson, 1957). They survive together in a community because they differ in the resources used or in the time of activity or place in the community. Niche differentiation, because it makes possible the coexistence of species in communities, is the basis of species diversity (Hutchinson, 1957). The three hypotheses set by Anderson (1975) in his 'Enigma of soil animal species diversity' can also be described as Hutchinson's niche theory.

The diversity of available resources is one determinant of the ecological diversification within a community. Communities with fewer different resources, and thus fewer niches, will support fewer species than those communities with a greater variety of resources (Pianka, 1975). As the diversity of utilization of resources by a species increases, its niche broadens, and the number of species that can coexist within a community must decrease (Pianka, 1975). In this way, an increase in the number of generalist species can limit resources for the specialist species (e.g. in agricultural systems).

Two communities can differ in species diversity if they differ in the average extent to which resources are shared, or the amount of niche overlap. Pianka (1978) suggests four hypothetical outcomes of different degrees of niche overlap. First, one niche might be totally included within a second, larger one. An inferior species would be eliminated and

the superior species would eliminate any other species that would contest the available space. Second, with partial overlap of niches each species has an exclusive, uncontested refuge. Coexistence is possible, yet depends on the amount of niche overlap. A community with greater overlap will support more species than one with less overlap simply because more species use each resource. Thirdly, niches may be touching each other. No direct competitive exclusion can occur, but such niche relations might reflect the avoidance of competition. Finally, niches are totally disjunct, such that both species occupy their own fundamental niche.

The niche concept facilitates further understanding of fundamental ecological questions such as what determines the abundance and number of species in a community (Schoener, 1989; Moore and de Ruiter, 1991)? What determines both the extent of species distributions and the variability from community to community in how particular species utilize resources (Schoener, 1989)? Most importantly we can ask, do the changes in the functional groups or species niche in human-modified ecosystems impact on basic ecosystem level processes such as decomposition and nutrient cycling? This conceptual framework is a topic of increasing focus in the literature and soil animals are now considered an appropriate level for investigation.

Those environments that are more spatially heterogeneous, having greater niche variety, can be expected to accommodate greater species richness, precisely because they provide a greater variety of microhabitats, a greater range of microclimates, more types of places to hide from predators and an increased resource spectrum (Begon *et al.*, 1996). Alternatively, one could hypothesize that the simplification of an ecosystem would lead to less spatial

heterogeneity, altered community structure and decreased species richness (Hansen and Coleman, 1998).

1.10 Research Aim

The primary aim of this study was to compare the soil and litter mite communities of two forest types, a *Eucalyptus globulus* plantation and an adjacent native dry sclerophyll (jarrah) forest. In Western Australia, it is common to reclaim pasture environments with these *E. globulus* plantations. Thus, there is an historical influence likely to predispose the establishment of the *E. globulus* plantations and as a result, a currently grazed pasture had to be included in the study. This enabled comparison of the mite communities in the sequence of grazed pasture to *E. globulus* plantation to native jarrah forest.

The specific objectives of the study were to:

- (i) determine the population abundance of mites in litter and soil layers in three vegetation systems in the Collie region of Western Australia; and
- (ii) determine the species richness and species diversity of the mite communities in the three systems.

These data were used to:

- (i) describe the taxonomic structure of the mite communities in forest, plantation and grazed pastures within the northern jarrah forest ecosystem;

- (ii) determine the potential impact of an introduced tree species monoculture, with its associated single-species litter layer, on the species composition and abundance of the mite communities; and
- (iii) describe the difference in mite abundances and diversity along the pasture to monoculture to forest habitat gradient in the Collie region of W.A..

Based on niche theory and litter differences it was hypothesized that:

H₁: The disturbance gradient, forest – plantation – pasture, will reflect differences in acarine species assemblages.

H₂: The monospecific nature of the litter in *E. globulus* plantations and the subsequent potential reduction in niche diversity, will reflect a reduced diversity of mite communities at these sites, in particular the fungivorous grazers.

CHAPTER 2: METHODOLOGY

2.1 The Study Region

The study sites are all situated in the region of Collie, Western Australia (Latitude 33° 21' S, Longitude 116° 08' E) (Figure 2.1). The region is part of the Avon Province, consisting of a gently undulating plateau commonly known as the Darling Scarp. Parent materials are of an older lateritic profile formed over deeply weathered granites and gneisses (Bettenay, 1983). The overlying lateritic soils are sandy-loam granite (Havel, 1975).

The Collie region forms the southern end of 10 500 km² of State Forest. Encompassing dry sclerophyll forest dominated by jarrah (*Eucalyptus marginata* Donn ex Sm.), the region is specifically known as the northern jarrah forest ecosystem (Havel, 1975) (Figure 2.1). It is an area renowned as a source of prime hardwood timber and contains catchments of many streams that supply the domestic, industrial and irrigation requirements of the southwest. Scattered through the area are exotic plantings of *Pinus* tree species and several varieties of *Eucalyptus* species. Large parts of the area are under long-term tenement for bauxite and coal mining. Most importantly, the northern jarrah forest ecosystem acts as a refuge for native fauna and flora threatened by clearing for human-modified environments (Dell, Havel and Malajcuk, 1989). The region experiences a mediterranean-type climate with dry, hot summers and cool, wet winters. Annual rainfall averages 800 - 1200 mm.

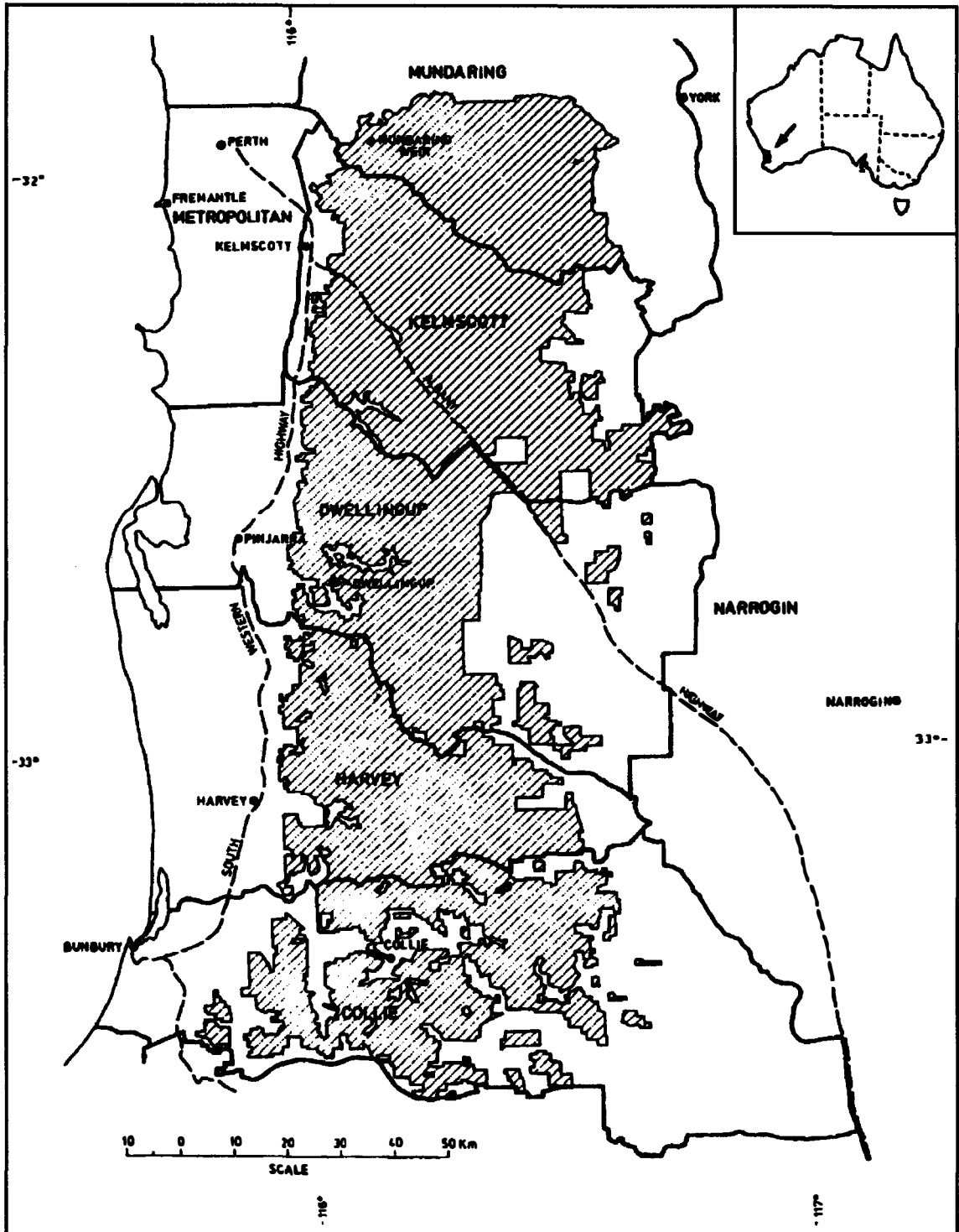


Figure 2.1. Locality of Collie, Western Australia, and the extent of the northern jarrah forrest ecosystem (Havel, 1975).

2.1.1 Native Forest

Three native forest areas were selected for sampling within a forest block (Fleays Block, covering 5602.4 ha) managed by the Department of Conservation and Land Management, Western Australia (Figure 2.2). This was the only forest block available which had relatively undisturbed vegetation and was adjacent to both the pasture and plantation ecosystems with a similar soil profile. The native vegetation of the area is dominated by jarrah with marri (*E. calophylla* R. Br. ex Lindl.) as the minor overstorey component (Hingston, Dimmock and Turton, 1980/1981). Other minor tree species include *Banksia*, *Acacia*, *Hakea*, *Allocasuarina* and *Xanthorrhoea* species. Annuals form a minor proportion of the vegetation.

The three areas were separated by access roads. Fleays Block was likely to be unaffected by land management techniques for the duration of the study. Fleays block suffered a fuel reduction burn by CALM in 1993, with another scheduled for 2002. Selective tree cutting occurred throughout the entire block prior to 1969.

2.1.2 *Eucalyptus globulus* Plantation

The three replicate plantation sites selected for sampling are situated in the plantation region known as Coolangatta and are managed by the Department of Conservation and Land Management's Sharefarmers Division for the Japanese owned Hansol Paper Pulp Company. Coolangatta covers 762.7 ha of reclaimed pasture of which *Eucalyptus globulus* plantings of various ages constitute 717.3 ha (Figure 2.3).

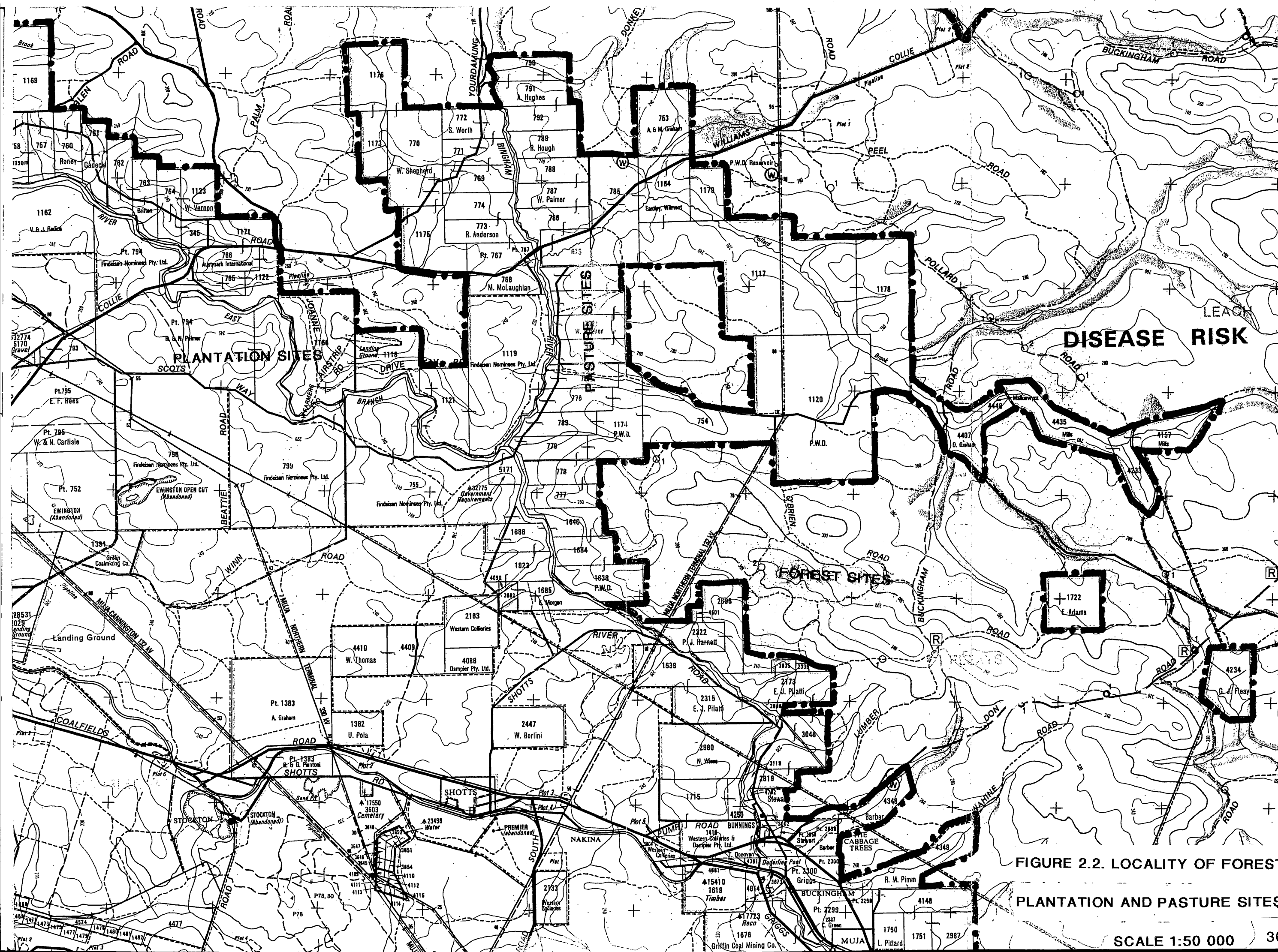


FIGURE 2.2. LOCALITY OF FOREST

PLANTATION AND PASTURE SITES

CALM SHAREFARMS LOWER WEST

COOLANGATTA

Wellington Loc. 755, 765, 766, 794, 799,
1118, 1119, 1121, 1166, 1171

PLANTATION PLAN LEGEND

	P.90 E.MUELLERANA G.P.S. CAPTURE		SEALED ROAD
	P.90 E.GLOBULUS G.P.S. CAPTURE		UNSEALED ROAD
	P.93 E.GLOBULUS G.P.S. CAPTURE		POWERLINE
	P.94 E.GLOBULUS G.P.S. CAPTURE		SWAMP
	P.95 E.GLOBULUS G.P.S. CAPTURE		DAM
	EXISTING BUSH G.P.S. Capture made tree crop area only		WATER POINT
	FENCE		BUILDINGS
	CADASTRAL BOUNDARY G.P.S. CAPTURE		CADASTRAL BOUNDARY NON G.P.S. CAPTURE

NB. RED FILLED EXCLUSION AREAS (25.7 HA)
NOT A PART OF PLANTATION TOTAL AREA

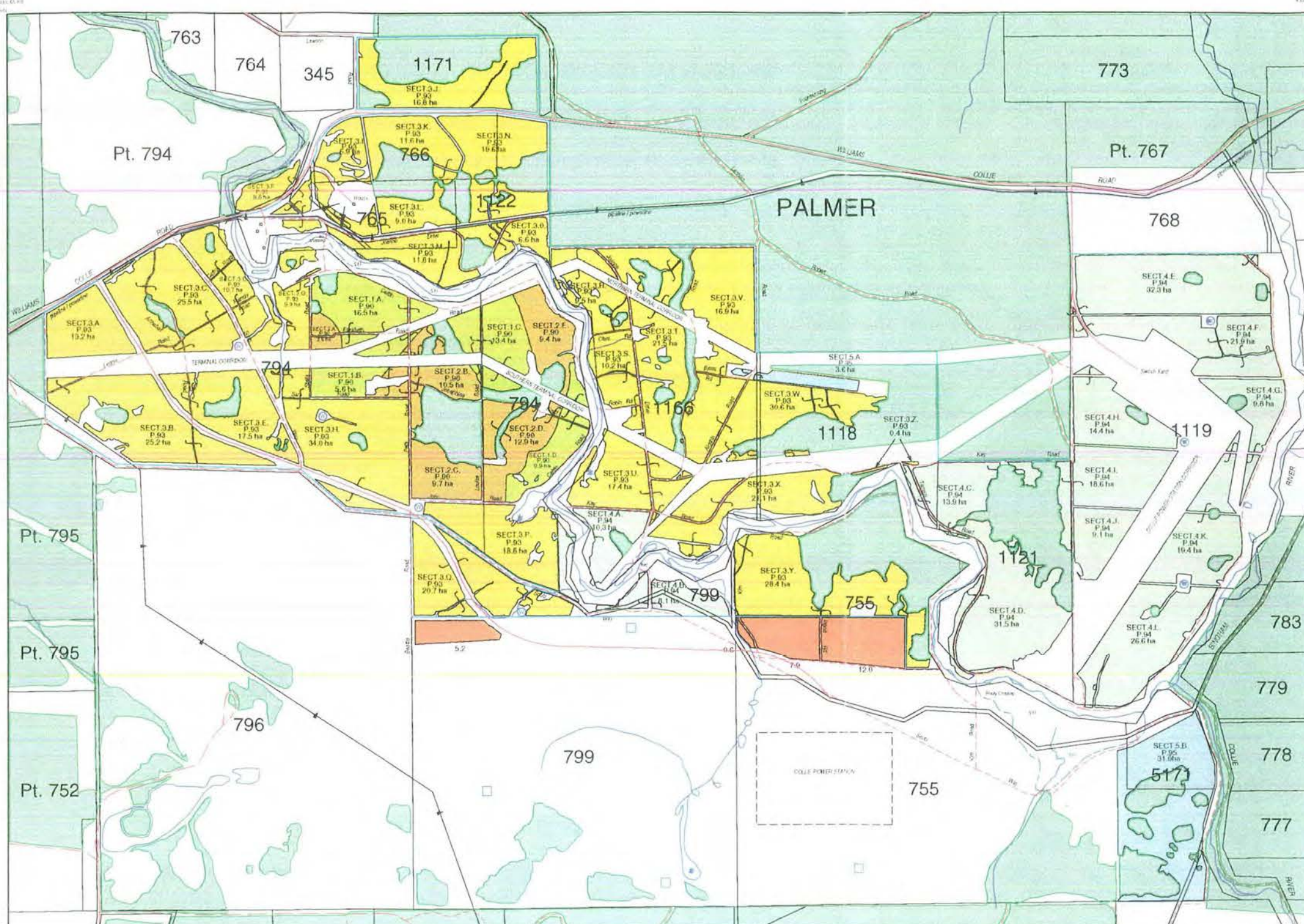
STATISTICAL REPORT

Categories	Area (ha)
P.90 E.MUELLERANA	45.4
P.90 E.GLOBULUS	45.1
P.93 E.GLOBULUS	421.7
P.94 E.GLOBULUS	215.9
P.95 E.GLOBULUS	34.6
TOTAL AREA	762.7

SHIRE: COLLIE
MAIN ACCESS ROAD: WILLIAMS COLLIE RD
FIRE CONTROL DETAILS
OWNER:
CONTACT:
FIRE CONTROL CONTACTS:
1.
2.
3.
FIREBREAKS
15m BOUNDARY
15m ADJOINING 1st CLASS ROADS
10m ADJOINING 2nd CLASS ROADS
6m INTERNAL BETWEEN COMPARTMENTS
6m INTERNAL

NB: The surrounding location boundaries have been determined solely by aerial mapping and are for information purposes only. There may be inconsistencies between the G.P.S. data and the cadastral data. It is important for the landholder to verify the location boundaries with the relevant authorities.

DEPARTMENTAL PLAN FOR OPERATIONAL USE ONLY



Part of CALM 1/50 000 map: COLLIE	Date: 27-11-95
Part of CALM 1/25 000 map: 2131-III-NE, 2131-II-NW	Date: 04-04-97
G.P.S. (Global Positioning System)	Date:
The Global Positioning System used is a real time G.P.S. which obtains accuracy of + 1.5m	
G.P.S. Surveyed By: S.Quain	
Plan Compiled By: C.Smith	
Plan Checked By:	

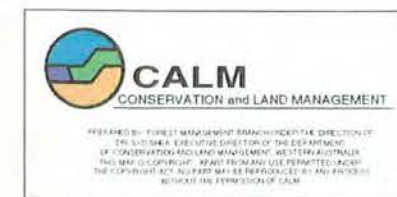
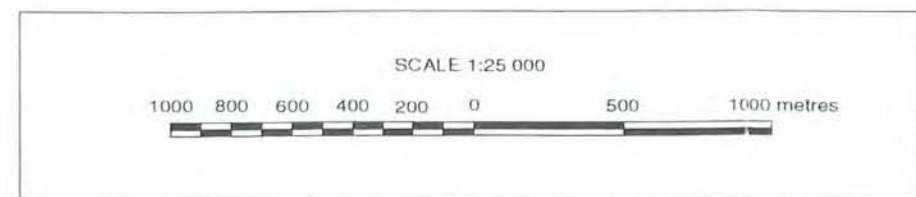


FIGURE 2.3. COOLANGATTA PLANTATION

Three *E. globulus* plantations of seven to eight years of age, the oldest plantations in the region, were chosen for sampling. Canopy closure was almost complete, and the troughs and furrows formed at establishment had disappeared. Although the crops were close to harvesting, they will not be harvested until the year 2003. These 1990 *E. globulus* plantations cover an area of 33.1 ha, with pockets of remnant jarrah forest dispersed throughout (Figure 2.3, P.90 *E. globulus* G.P.S. Capture). At the time of planting, each *E. globulus* tree received 50 g of diammonium phosphate (DAP) fertilizer. In 1997, established plantings received 100 kg ha⁻¹ of urea. This was a blanket application to increase concentrations of nitrate and phosphate.

2.1.3 Grazed Pasture Land

The three pasture sites selected for sampling are situated on Myaree Farm, adjacent to both Coolangatta and Fleays Block. The pastures of the property are currently being grazed at the stocking level of three sheep per acre (0.405 ha). Ryegrass (*Lolium sp.*) and subterranean clover (*Trifolium subterraneum*) predominantly cover 405 ha of the property. Superphosphate and Potash fertilizers are applied in a ratio of 3:1 at 125 kg ha⁻¹ annually. The only herbicide used is for firebreak maintenance; sampling occurred well away from the firebreaks.

2.2 Sampling Methods

The time required to process and analyse samples to the morpho-species level has been referred to in section 1.6. The study design therefore needed to take into account this time demand. Sampling of the litter and soil was carried out in spring (mid-October 1997) and at newly selected sites in late autumn (May 1998). These periods were chosen to correspond with peak abundance and activity in the soil and litter system (Postle, 1989). Samples were always taken between the hours of 0600 and 1100.

In each of the nine sampling sites (3 plantation, 3 pasture and 3 forest), one quadrat (10m x 10m square) was randomly placed for the sampling of soil and litter mesofauna. Within the quadrat, twelve sample points were randomly selected. At two of these sample points approximately 3 litre volumes of litter were removed and sealed in a plastic bag. A soil core (5cm diameter x 10cm depth) was taken at each of the remaining ten sample points, following the removal of litter. Thus, a total of 3 x 2 litter samples and 3 x 10 soil samples were taken from the two forest vegetation types, on each date. There was no litter in the pasture sites.

At each sampling site, the following variables were measured:

(i) Temperature: Immediately prior to sampling, temperatures were taken at the litter surface, the soil surface and at a soil depth of 10 cm.

(ii) Vegetation Mapping: Maps showing the positions of major tree and shrub species were made for each of the sampling sites within the native forest. Vegetation strata

classifications were based on Specht (1970; 1981), defining vegetation structures as tallest stratum, mid-stratum (understorey) and lower stratum (ground layer). In the native forest and *E. globulus* plantation sites litter depth was measured and litter samples were described by the percentage weight of leaves, sticks and other debris, after faunal extraction.

Soil was collected from each sampling site for the following measurements to be performed in the laboratory:

(i) Soil Moisture: For soil moisture analysis 2 x 50 g samples of soil were oven dried at 50 - 60°C to constant weight and the water content expressed as a percentage of the total dry weight.

(ii) Soil Particle Size: This was determined on the dried soil cores after animal extraction. Large sticks and stones were removed before the soil was passed through a standard set of soil sieves to estimate the classes of particle size. Particle size classifications were based on those classifications set out by the Standards Association of Australia (McDonald, Isbell, Speight, Walker and Hopkins, 1990).

(iii) pH: Soil samples were obtained from the top 1 cm of the soil strata. A ratio of 10g of soil to 100g of distilled water was stirred for 24 hours prior to measuring with a CyberScan pH 500 meter.

Rainfall: Precipitation measurements were obtained from the Bureau of Meteorology.

Measurements were recorded at Wokalup Agricultural Research Station located at Harvey, Western Australia, approximately 70 km from the study sites.

2.3 Faunal Extraction

All samples were immediately returned to the laboratory and the soil and litter fauna extracted using a method based on that of Kempson, Lloyd and Ghelardi (1963) modified to process either litter samples or soil core samples. The extraction method utilizes a moisture and temperature gradient to expel fauna from the litter and soil samples. The extractions ran over a ten-day period during which the temperature of the samples was raised from room temperature to approximately 50°C. This produced an effective temperature gradient of 25 - 30°C from top to bottom of the sample. Empty control cores were also placed in the extractors to test for presence of non-soil invaders in the extracted soil samples. No animals were found in these cores.

2.4 Sample Processing

Mites from the soil and litter samples were sorted under a dissecting microscope (magnification x 40), counted and classified to species or morphospecies where appropriate. All reference to species or morphospecies in this study are hereafter called species. A voucher collection of mounted specimens was developed together with a reference catalogue including descriptions, photographs and detailed drawings of differentiating characteristics. The voucher collection is maintained at the School of Natural Sciences, Edith Cowan University, Western Australia. Abundance of all other faunal groups, such as Collembola and Insecta, were determined at a higher taxon level.

2.5 Statistical Analyses

The data were initially entered into a species abundance-by-site matrix using the Excel computer package. Species were coded to allow easy selection and analyses at various taxon levels. All immature mites were excluded from all analyses due to taxonomic difficulties and time constraints.

The forest sites, though clearly separated by access roads, were within one forest block. While it could be argued that, given the scale of mite habitat relative to site distances, the sites were replicates, in this study a conservative view has been taken in regard to site replication. These sites are considered to be pseudoreplicates rather than true replicates. For this reason the study has a pseudoreplicate design, and is thus limited in the use of parametric statistical techniques (Hurlbert, 1984). Statistical analyses were carried out using SPSS. Descriptive statistics such as the mean, sum, and standard error of site data were computed. Results of litter samples from each replicate quadrat within a site were combined and animal abundance was expressed as abundance kg^{-1} dry weight of litter, giving 3 replicate values per vegetation type. Abundance in each soil core (10 per quadrat) was expressed as numbers m^{-2} , which is consistent with the majority of soil faunal studies. In both cases, data transformation of abundances to $\text{Ln}(n+1)$ was sufficient to normalize the data and provide homoscedacity.

Traditional community attributes such as species diversity and ecological similarity of species compositions were computed using Krebs Ecological software (Krebs, 1995).

Species diversity was measured with Simpson's reciprocal index ($1/D$) as recommended by Krebs (1989). Morisita's similarity index was used to measure the degree of overlap of component species between communities (Krebs, 1989). Rank-abundance (Whittaker) plots, a logarithmic heterogeneity measure, were constructed to compare community structure between sites.

PATN software (Belbin, 1989) was used for ordination and classification procedures to identify major structural patterns of species distributions in replicates, and to compare the species complement of sites. The ordination procedure used Semi-Strong Hybrid (SSH) multidimensional scaling. Sites most similar in both species composition and relative abundance appeared closest together, whilst sites that differed greatly in the relative importance of a similar set of species, or that possess quite different species, appear far apart. The Hierarchical Agglomerative clustering (classification) analysis utilized the procedure UPGMA using the association measure, Kulczynski's coefficient. Sampling sites with similar species compositions were grouped in subsets based on the resultant degree of dissimilarity.

CHAPTER 3: RESULTS

3.1 Vegetation Analyses

3.1.1 Native Forest Sites

The native forest used in this study is classified as open forest based on a canopy density of 30-70% crown cover (Havel, 1975; Specht, 1981). The sample sites varied in understorey plant components reflecting variations in soil and drainage patterns.

Spring site 1 (Plate 3.1a): The canopy vegetation (tallest stratum) was exclusively *Eucalyptus calophylla* (marri). Within the understorey (mid-stratum), *Hakea lissocarpha* (honey bush) was present in high numbers. The ground layer (lower stratum) was a mixture of *Bossiaea pulchella*, *Goodenia caerulea*, *Haemodorum sp.*, *Hibbertia sp.*, *Leucopogon capitellatus*, and *Synaphea sp.* Litter was composed mainly of eucalypt leaves (61%), with sticks, twigs and nuts making up the remainder (39%). Litter depth ranged between 0.5 and 1.5 cm.

Spring site 2 (Plate 3.1b): In this site, the canopy vegetation was exclusively *Eucalyptus marginata* (jarrah). The understorey was composed of *Banksia grandis* (bull banksia) and *Allocasuarina humilis* (scrub sheoak). The ground layer was of *B. pulchella* and *Hibbertia sp.* Litter was predominantly leaves (63%), with sticks, twigs and nuts (37%). Litter depth ranged between 1.0 and 3.0 cm.



Plate 3.1a. Native forest site 1, spring 1997.



Plate 3.1b. Native forest site 2, spring 1997.

Spring site 3 (Plate 3.1c): A canopy of *E. marginata* was again present with a dense understorey of *H. lissocarpha*, *Macrozamia riedlei* (zamia cycad), *Xanthorrhoea sp.* (grass tree) and *Acacia pulchella* (prickly moses). The ground layer included *B. pulchella*, and myrtle *sp.*. There were equal proportions of leaves and sticks, twigs and nuts in the litter. Litter depth ranged between 1.0 and 4.0 cm.

Autumn site 1 (Plate 3.2a): The canopy vegetation was *E. marginata* with an understorey of *B. grandis*, *A. humilis*, *H. lissocarpha* and *M. riedlei*. The ground layer included *Hibbertia sp.* and *L. capitellatus*. There were equal proportions of leaves and sticks, twigs and nuts in the litter which had a depth ranging between 1.0 and 2.0 cm. Large fallen logs were distributed over the area.

Autumn site 2 (Plate 3.2b): *E. marginata* was the canopy vegetation with a mixed understorey of *A. pulchella*, *A. humilis*, *M. riedlei* and *Persoonia longifolia* (snottygobble). The ground layer included *Hibbertia sp.*. Litter was composed predominantly of leaves (60%), with sticks, twigs and nuts (40%). Litter depths ranged between 2.0 and 3.0 cm. Fallen logs were distributed over the area.

Autumn site 3 (Plate 3.2c): A canopy of *E. marginata* was again present with an understorey composed of *H. lissocarpha*, *M. riedlei* and *Xanthorrhoea sp.*. The ground layer included *Hibbertia sp.*. Equal proportions of leaves and sticks, twigs and nuts comprised the litter. Litter depth ranged between 1.0 and 2.0 cm.



Plate 3.1c. Native forest site 3, spring 1997.



Plate 3.2a. Native forest site 1, autumn 1998.



Plate 3.2b. Native forest site 2, autumn 1998.



Plate 3.2c. Native forest site 3, autumn 1998.

Pictorial vegetation maps describing the distribution of the major vegetation component in each sample quadrat of the native forest sampling sites are in Appendix I.

3.1.2 *Eucalyptus globulus* Plantation Sites

Within the *Eucalyptus globulus* plantations, tree stands were similar at each sample site. The touching crown cover classifies the bluegum plantations as closed, and no other plant species were present. Litter composition was uniform throughout the sampling period and comprised 50 - 65% *E. globulus* leaves and 35 - 50% *E. globulus* bark, sticks and twigs (Plate 3.3a). Litter depth varied between 1.0 and 8.0 cm with depth being greater at the base of trees. There were numerous branches and several felled trees at each of the sites (Plate 3.3b).

3.1.3 Pasture Sites

Ryegrass (*Lolium sp.*) and subterranean clover (*Trifolium subterraneum*) were the predominant pasture vegetation with some dandelion and corkscrew (*Stipa sp.*).

During the sampling periods plant growth in the pastures varied. In spring, plant height ranged between 5.0 and 20.0 cm (Plate 3.4a). In autumn, plant height ranged between 2.0 and 10.0 cm (Plate 3.4b). Sheep continuously grazed the pastures. Sheep dung was deposited throughout the sampling sites in both seasons.



Plate 3.3a. An 8 year-old *E. globulus* plantation site. Note the fallen branches and build-up of litter at the base of trees.



Plate 3.3b. Typical composition of surface litter in an 8 year-old *E. globulus* plantation.



Plate 3.4a. Pasture site, spring 1997.



Plate 3.4b. Pasture site, autumn 1998.

3.2 Environmental Variables

3.2.1 Soil pH

Topsoil pH was below neutral at all three sites in both spring and autumn (Table 3.1). There was little seasonal variation in the native forest and plantation sites, but the pasture was lower in pH in autumn. The *E. globulus* plantation had the lowest values in both seasons.

Table 3.1. Mean pH of soil in native *E. marginata* forest, 8 year-old *E. globulus* plantation and grazed pasture sites.

Sample Period	Site	Soil pH Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	6.2 ± 0.20 (3)
	<i>E. globulus</i> plantation	5.7 ± 0.08 (3)
	Pasture	6.1 ± 0.07 (3)
Autumn 1998	<i>E. marginata</i> forest	6.3 ± 0.07 (3)
	<i>E. globulus</i> plantation	5.6 ± 0.11 (3)
	Pasture	5.7 ± 0.03 (3)

3.2.2 Soil Particle Size

Particle size distributions of topsoil from the sampling sites were typical of the region (Havel, 1975). The dominant fractions in the native forest and plantation sites were a mixture of coarse and fine gravel with traces of sand and coarse silt (Figure 3.1). In the pasture sites, fine gravel alone was the dominant fraction.

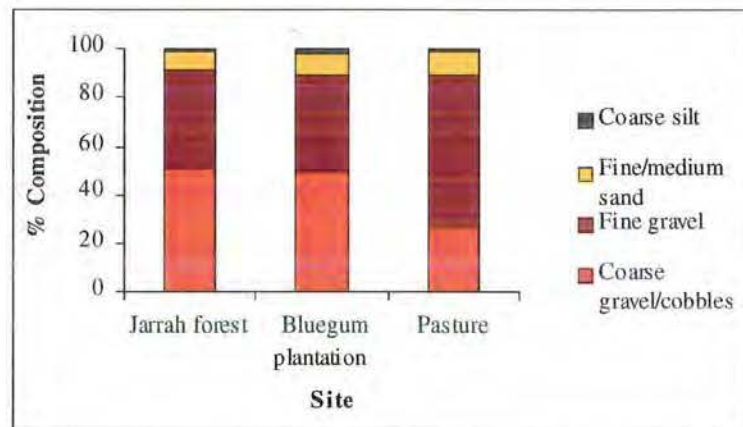


Figure 3.1. Particle size of soils from the *E. marginata* forest, *E. globulus* plantation and pasture sites.

3.2.3 Temperature

Ambient temperatures recorded during the sampling periods ranged between 14 - 20 °C in spring and 7 - 18 °C in autumn (Figures 3.2 and 3.3). Mean temperatures recorded at the surface litter and soil was consistently higher in spring than in autumn.

In spring, when the ambient temperatures were the highest, the native forest sites exhibited greater temperature gradients from the litter to the 10 cm soil depth (approximately 8 °C),

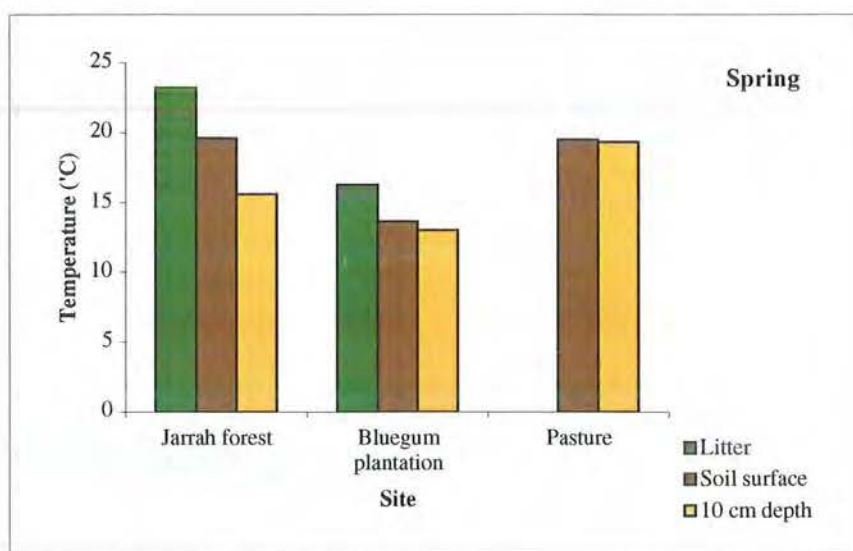


Figure 3.2. Mean temperatures of surface litter, soil surface and at 10 cm soil depth in *E. globulus* plantation, native *E. marginata* forest and pasture sites during sampling in spring 1997.

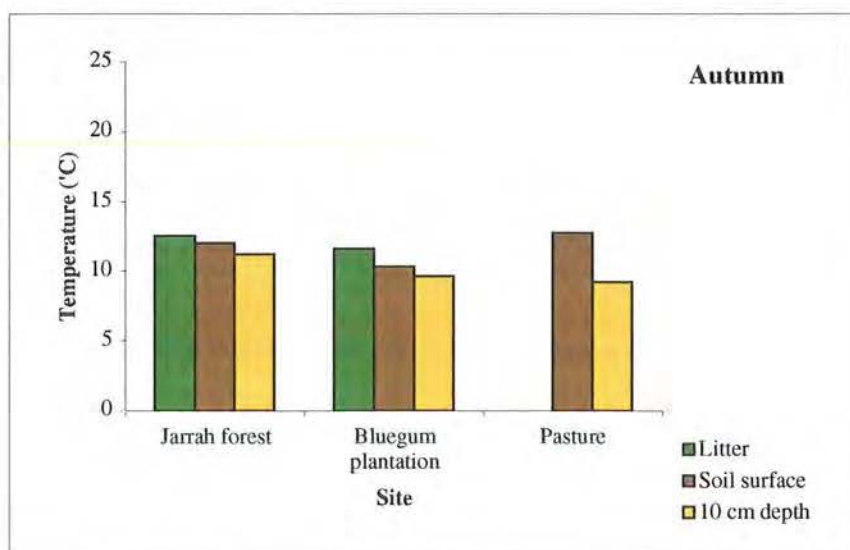


Figure 3.3. Mean temperatures of surface litter, soil surface and at 10 cm soil depth in *E. globulus* plantation, native *E. marginata* forest and pasture sites during sampling in autumn 1998.

due mainly to the higher temperatures at the litter and soil surface. The full canopy cover of the *E. globulus* plantation reduced the litter and soil surface temperatures, thus a less marked temperature gradient from the litter to the 10 cm soil depth. The pasture, with no litter to insulate the soil from the effects of the climate, had high temperatures at the soil surface and 10 cm soil depth.

In autumn, the native forest and *E. globulus* plantation were similar in temperature gradient, due to a more marked effect of the canopy cover. The soil surface temperature in the pasture was high, yet the temperature remained low at the 10 cm soil depth due to the reduced ambient temperatures and length of sunlight hours reaching the soil.

3.2.4 Rainfall

No rainfall was recorded during the spring sampling period (2 days in mid-October), however 24 mm of rain was recorded two days prior to sampling at approximately 70 km from the sampling sites (Bureau of Meteorology, Harvey, 1997-1998). The spring sampling occurred during a period of low rainfall (Figure 3.4).

During the autumn sampling period (2 days in late May) a large thunderstorm resulted in 70.6 mm of rain falling in the area over three days. Prior to this, rain had fallen consistently throughout the region over a period of a week. The autumn sampling period occurred during a period of increasing rainfall (Figure 3.4).

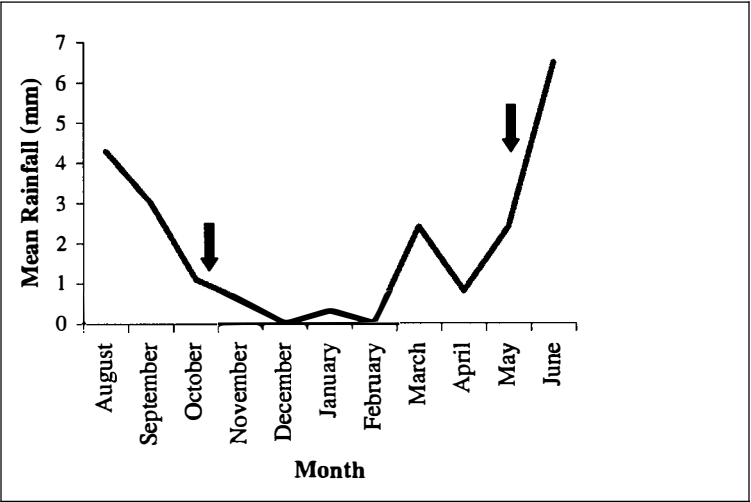


Figure 3.4. Mean monthly rainfall (mm) recorded for August 1997 through to June 1998.
Recorded at Wokalup Agriculture Research Station, Harvey (Bureau of Meteorology).
Arrows indicate sampling times.

3.2.5 Soil Moisture

Soil moisture values were influenced by rainfall preceding or during the sampling periods. All sites had the highest soil moisture levels in autumn (Table 3.2) reflecting the higher rainfall at this time. The pasture sites had the highest soil moisture values in both seasons.

Table 3.2. Mean moisture percentages of soil in native *E. marginata* forest, 8 year-old *E. globulus* plantation and grazed pasture sites.

Sample Period	Site	Soil Moisture (% dry weight) Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	13.0 ± 1.53 (3)
	<i>E. globulus</i> plantation	11.3 ± 1.16 (3)
	Pasture	13.5 ± 6.86 (3)
Autumn 1998	<i>E. marginata</i> forest	20.6 ± 1.86 (3)
	<i>E. globulus</i> plantation	15.3 ± 6.98 (3)
	Pasture	16.2 ± 6.91 (3)

3.3 Mesofaunal Abundance in Surface Litter

The numbers of mesofauna extracted from the surface litter at both the *E. globulus* plantation and native forest sites were considerably greater in autumn than in spring by a factor of ten (Table 3.3). Within seasons, mean abundance values were similar between the two sites.

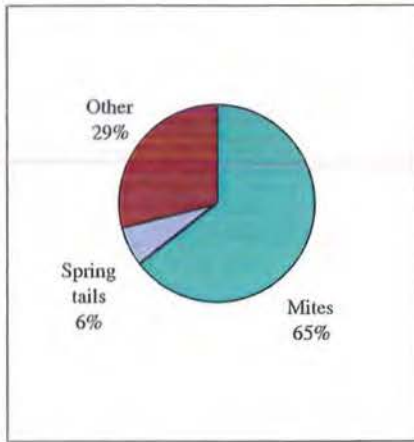
Table 3.3. Mean abundance (No. kg⁻¹ dry litter) of mesofauna in surface litter of a native *E. marginata* forest and three 8 year-old *E. globulus* plantations.

Sample period	Site	Mesofaunal abundance (No. kg ⁻¹ dry litter)
		Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	291 ± 63.6 (3)
	<i>E. globulus</i> plantation	178 ± 67.1 (3)
Autumn 1998	<i>E. marginata</i> forest	2865 ± 700.1 (3)
	<i>E. globulus</i> plantation	2920 ± 689.6 (3)

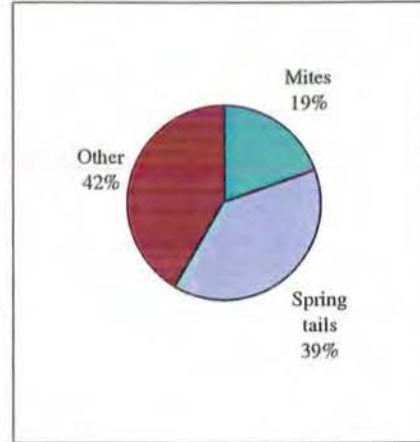
With one exception (the spring *E. globulus* plantation), mites were the numerically dominant faunal group, contributing well over 60 % of the total numbers extracted (Figures 3.5 and 3.6). The springtails contributed higher percentages in the *E. globulus* plantation sites.

The faunal component ‘Other’ (the non-mite and non-springtail fauna) was relatively large (29 – 42 %) in the spring samples from both sites, yet it only comprised 3 % of the autumn fauna. ‘Other’ animals included centipedes (Chilopoda), millipedes (Diplopoda), pseudoscorpions (Pseudoscorpionidea), spiders (Araneida), diplurans (Diplura),

Spring



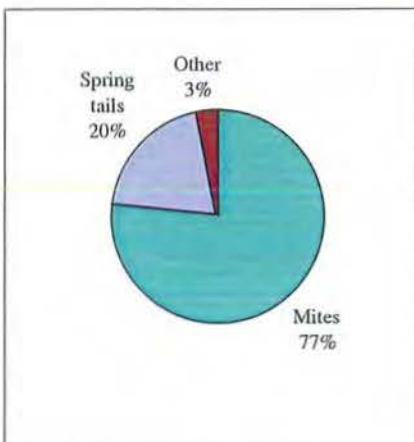
Native *E. marginata* forest



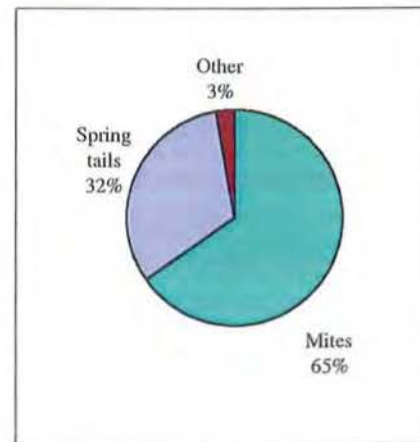
E. globulus plantation

Figure 3.5. Percentage contributions of each mesofaunal group in surface litter of a native *E. marginata* forest and three 8 year-old *E. globulus* plantation sites in spring 1997.

Autumn



Native *E. marginata* forest



E. globulus plantation

Figure 3.6. Percentage contributions of each mesofaunal group in surface litter of a native *E. marginata* forest and three 8 year-old *E. globulus* plantation sites in autumn 1998.

earwigs (Dermaptera) and ants (Formicidae). Of these, ants were the most dominant (50 – 60 %).

3.3.1 Acarine Abundance in Surface Litter

Seasonal differences in acarine mean abundance (Table 3.4) mirrored those of the mesofauna generally, with substantially more mites extracted from the autumn samples from both vegetation sites. The seasonal difference was particularly marked in the *E. globulus* samples due to very low numbers extracted in spring. There were consistent site differences, with a trend of larger abundances in the native forest litter.

Table 3.4. Mean acarine abundance (No. kg⁻¹) in surface litter of a native *E. marginata* forest and three 8 year-old *E. globulus* plantations.

Sample period	Site	Acarine abundance (No. kg ⁻¹ dry litter) Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	194 ± 19.2 (3)
	<i>E. globulus</i> plantation	34 ± 12.3 (3)
Autumn 1998	<i>E. marginata</i> forest	2193 ± 447.7 (3)
	<i>E. globulus</i> plantation	1867 ± 601.4 (3)

The greater acarine abundances in autumn were consistently reflected at all four ordinal levels (Table 3.5; Figures 3.7 and 3.8).

Table 3.5. Mean abundance (No. kg⁻¹) of acarine orders in surface litter of a native *E. marginata* forest and an 8 year-old *E. globulus* plantation.

Sample period	Site	Mesostigmata (No. kg ⁻¹ dry litter) Mean ± SE (N)	Prostigmata (No. kg ⁻¹ dry litter) Mean ± SE (N)	Cryptostigmata (No. kg ⁻¹ dry litter) Mean ± SE (N)	Astigmata (No. kg ⁻¹ dry litter) Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	12 ± 4.7 (3)	30 ± 2.5 (3)	145 ± 16.9 (3)	7 ± 4.9 (3)
	<i>E. globulus</i> plantation	1 ± 1.1 (3)	18 ± 4.2 (3)	2 ± 1.2 (3)	13 ± 10.5 (3)
Autumn 1998	<i>E. marginata</i> forest	464 ± 209.9 (3)	56 ± 9.1 (3)	434 ± 277.9 (3)	1239 ± 140.9 (3)
	<i>E. globulus</i> plantation	431 ± 250.0 (3)	88 ± 12.4 (3)	62 ± 12.4 (3)	1286 ± 337.8 (3)

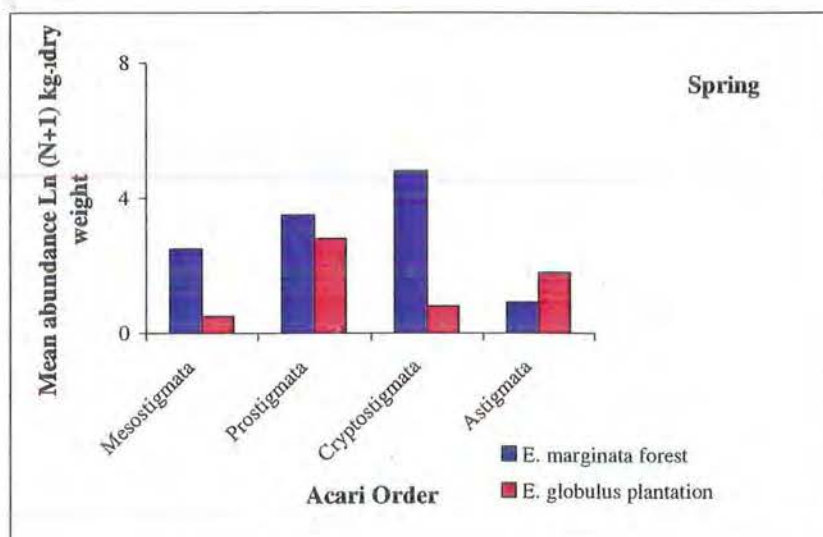


Figure 3.7. Mean abundance (expressed as natural log) of acarine orders in surface litter of a native *E. marginata* forest and three 8 year-old *E. globulus* plantations in spring 1997.

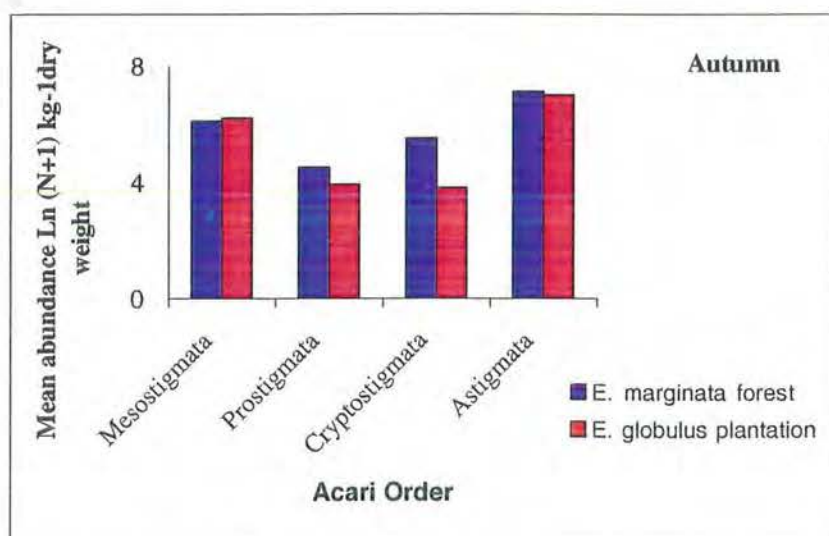


Figure 3.8. Mean abundance (expressed as natural log) of acarine orders in surface litter of a native *E. marginata* forest and three 8 year-old *E. globulus* plantations in autumn 1998.

Mesostigmata and Astigmata showed the greatest increase in abundance between the 1997 and 1998 sample periods, especially in the *E. globulus* plantation. The Oribatida were considerably more abundant in the native forest, particularly in spring. Similar, though less marked trends were seen with the Prostigmata. On the other hand, the Astigmata were always relatively well represented in the *E. globulus* plantation.

3.3.2 Collembolan Abundance

Like the mites, the numbers of springtails extracted from surface litter were significantly greater in autumn than in spring (Table 3.6). Abundances were consistently higher in the *E. globulus* plantation sites.

Table 3.6. Mean collembolan abundance (No. kg⁻¹) in surface litter of a native *E. marginata* forest and three 8 year-old *E. globulus* plantations.

Sample period	Site	Collembolan abundance (No. kg ⁻¹ dry litter)
		Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	17 ± 11.8 (3)
	<i>E. globulus</i> plantation	70 ± 52.5 (3)
Autumn 1998	<i>E. marginata</i> forest	579 ± 306.2 (3)
	<i>E. globulus</i> plantation	972 ± 210.9 (3)

3.4 Mesofaunal Abundance in the Soil

The mean numbers of mesofauna extracted from the soils at each of the three sites were greater in autumn than in spring (Table 3.7). Seasonal differences were particularly marked in the pasture sites, where very high numbers were extracted in autumn. The native forest and *E. globulus* plantation sites had similar abundances in both seasons.

Table 3.7. Mean abundance (No. m⁻²) of mesofauna in soil under a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and a grazed pasture.

Sample period	Site	Mesofaunal abundance x 10 ³ (No. m ⁻²)
		Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	74 ± 13.2 (3)
	<i>E. globulus</i> plantation	75 ± 19.6 (3)
	Pasture	41 ± 12.8 (3)
Autumn 1998	<i>E. marginata</i> forest	114 ± 3.4 (3)
	<i>E. globulus</i> plantation	195 ± 23.7 (3)
	Pasture	492 ± 23.3 (3)

The trends in percentage contributions of each mesofaunal group extracted (mites, springtails and ‘Other’) were similar between sites for both sampling periods, with the exception of spring pasture (Figures 3.9 and 3.10). Unlike the litter samples, the total percentage contribution of the ‘Other’ faunal component was relatively small and there was less diversity at the higher taxon level, in which ants and centipedes were numerically dominant. The springtails were always low in abundance, with the exception of spring pasture. Mites were the most numerically dominant soil mesofauna, contributing over 86 % of the total numbers.

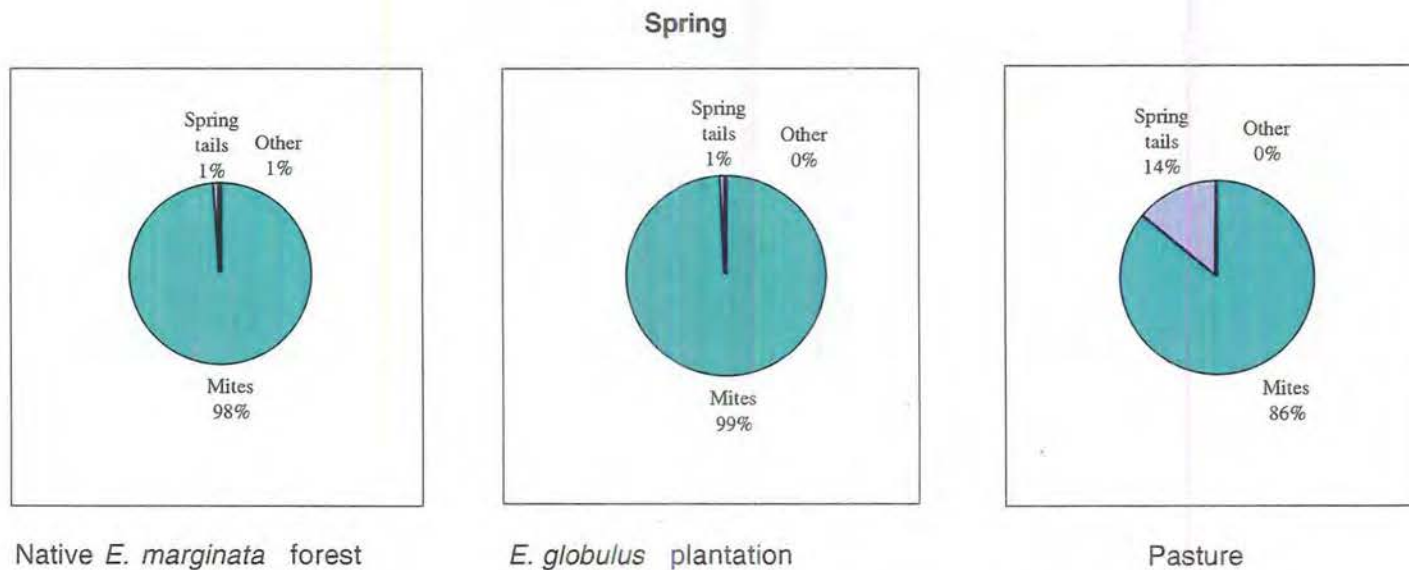


Figure 3.9. Percentage contributions of each mesofaunal group in soil under a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and a grazed pasture in spring 1997.

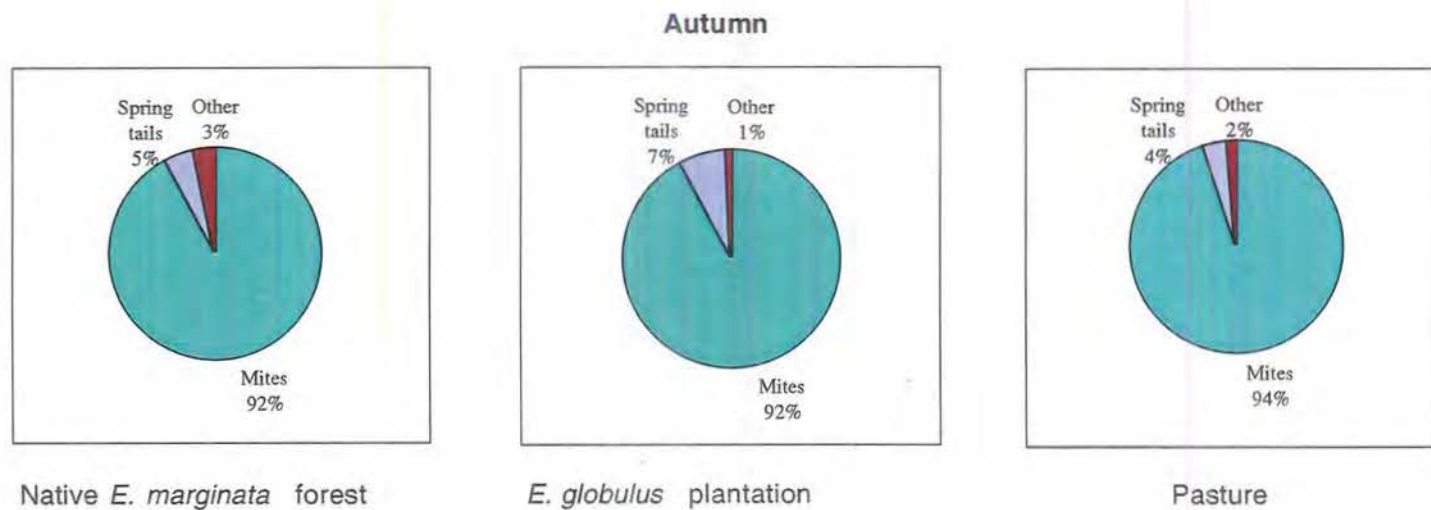


Figure 3.10. Percentage contributions of each mesofaunal group in soil under a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and a grazed pasture in autumn 1998.

3.4.1 Acarine Abundance in the Soil

Seasonal differences in soil acarine mean abundance mirrored that of the soil and litter mesofauna generally (Table 3.8), with substantially more mites extracted from the autumn samples. This mirroring is expected, given that the mites contribute to the greater part of the soil fauna. Abundances in the *E. globulus* plantation sites were slightly higher than in the native forest at both sampling periods. Seasonal differences in the pasture sites were marked due to very low numbers extracted in spring and the large increase in Astigmata abundances in autumn.

Table 3.8. Mean acarine abundance (No. m⁻²) in soil under a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and a grazed pasture.

Sample period	Site	Acari abundance x 10 ³ (No. m ⁻²)
		Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	73 ± 12.8 (3)
	<i>E. globulus</i> plantation	75 ± 19.5 (3)
	Pasture	35 ± 15.3 (3)
Autumn 1998	<i>E. marginata</i> forest	105 ± 4.4 (3)
	<i>E. globulus</i> plantation	181 ± 27.1 (3)
	Pasture	467 ± 32.5 (3)

The seasonal pattern observed for total mite abundance was reflected in the four acarine orders (Table 3.9; Figures 3.11 and 3.12).

Table 3.9. Mean abundance (No. m⁻²) of acarine orders in soil under a native *E. marginata* forest, an 8 year-old *E. globulus* plantation and grazed pasture.

Sample period	Site	Mesostigmata (No. x 10 ³ m ⁻²) Mean ± SE (N)	Prostigmata (No. x 10 ³ m ⁻²) Mean ± SE (N)	Cryptostigmata (No. x 10 ³ m ⁻²) Mean ± SE (N)	Astigmata (No. x 10 ³ m ⁻²) Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	0.1 ± 0.06 (3)	0.3 ± 0.07 (3)	0.4 ± 0.14 (3)	72 ± 12.8 (3)
	<i>E. globulus</i> plantation	0.2 ± 0.09 (3)	0.2 ± 0.07 (3)	0.1 ± 0.07 (3)	74 ± 19.5 (3)
	Pasture	0.2 ± 0.02 (3)	0.1 ± 0.05 (3)	0.0 (3)	35 ± 15.3 (3)
Autumn 1998	<i>E. marginata</i> forest	28 ± 3.0 (3)	4 ± 0.8 (3)	5 ± 1.5 (3)	67 ± 6.0 (3)
	<i>E. globulus</i> plantation	41 ± 6.6 (3)	2 ± 0.4 (3)	12 ± 4.0 (3)	125 ± 24.0 (3)
	Pasture	108 ± 15.3 (3)	2 ± 0.3 (3)	0.0 (3)	357 ± 45.0 (3)

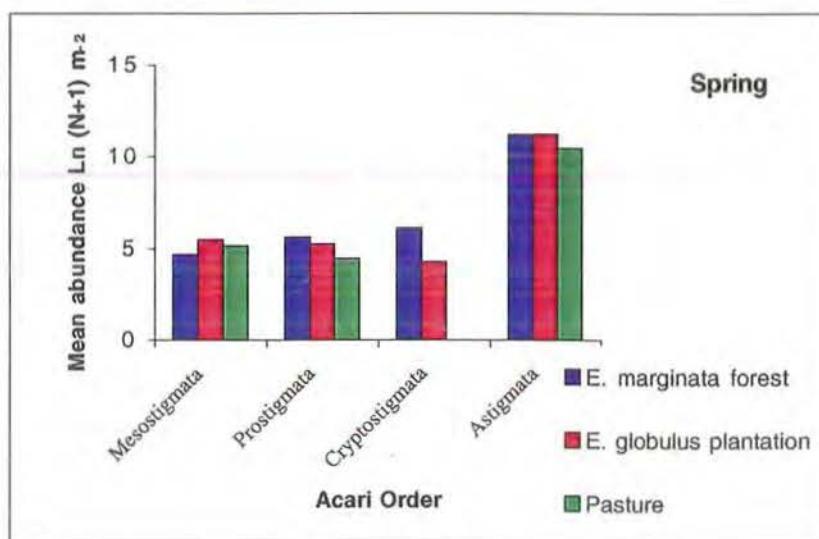


Figure 3.11. Mean abundance (expressed as natural log) of acarine orders in soil under a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and grazed pasture in spring 1997.

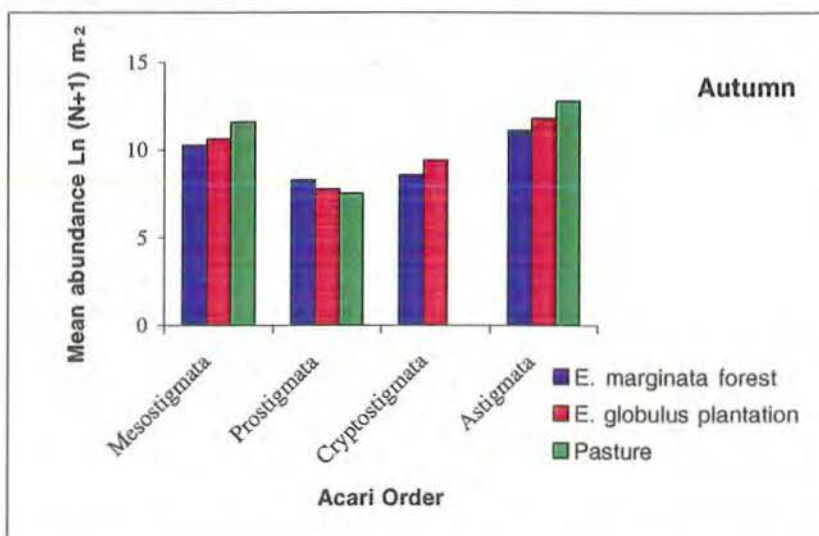


Figure 3.12. Mean abundance (expressed as natural log) of acarine orders in soil under a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and grazed pasture in autumn 1998.

The Mesostigmata were well represented in the soil from all three sites, with no obvious site differences, following a similar trend to the Astigmata in autumn. The Prostigmata were also well represented, yet were the least abundant order in autumn. A noteworthy result of the Oribatida is that they were totally absent from the pasture samples and were in lower numbers in the drier *E. globulus* plantation samples. The Astigmata were found at all three sites and in consistently very large numbers.

3.4.2 Collembolan Abundance

Like the mites, the number of springtails extracted from the soils was greater in autumn than in spring (Table 3.10). Abundances in the pasture sites were consistently higher than the native forest and *E. globulus* plantation, particularly in the spring samples.

Table 3.10. Mean Collembolan abundance (No. m⁻²) in soil under a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and a grazed pasture.

Sample period	Site	Collembolan abundance x 10 ³ (No. m ⁻²)
		Mean ± SE (N)
Spring 1997	<i>E. marginata</i> forest	0.6 ± 0.3 (3)
	<i>E. globulus</i> plantation	0.6 ± 0.1 (3)
	Pasture	5.6 ± 2.5 (3)
Autumn 1998	<i>E. marginata</i> forest	4.9 ± 0.5 (3)
	<i>E. globulus</i> plantation	13.2 ± 4.0 (3)
	Pasture	17.3 ± 9.5 (3)

3.5 Consideration of the Acarine Fauna at the Species Level

A total of 114 mite species was recognized; 16 Mesostigmata, 52 Prostigmata, 45 Cryptostigmata and 1 Astigmata. Of the total mite species, 44 species were recorded only in the soil, 30 species only in the surface litter, and 40 species in both the surface litter and soil. The full list of mite species extracted is in Appendix II.

3.5.1 Acarine Species Richness

Seasonal and site factors influenced species richness in both the surface litter and soil. At the seasonal level overall species richness was consistently higher in autumn than in spring (Table 3.11). Each acarine order was higher in species richness in autumn (with the exception of Prostigmata in the *E. globulus* litter) due to the greater abundances recorded in this season. The absence of Mesostigmata from all the drier spring samples, with the exception of the one species in the native forest soil, demonstrates the consequence of lower species richness from lower abundances. Figures 3.13 and 3.14 illustrate the relationship between mite abundance and species richness. In the surface litter (Figure 3.13), the relationship between increasing abundance and increasing species richness is strong in the native forest, yet less marked in the *E. globulus* plantation. The soil sites (Figure 3.14) illustrate no strong relationship between increasing abundance and increasing species richness. The single Astigmata genus, *Tyrophagus sp.1*, was present in the soil and litter layers of all three sites in relatively high abundances.

Table 3.11. Species richness of acarine orders in the surface litter and soil of a native *E. marginata* forest, an 8 year-old *E. globulus* plantation and grazed pasture in the sampling seasons of spring 1997 and autumn 1998.

Acarine order	<i>E. marginata</i> forest				<i>E. globulus</i> plantation				Pasture	
	Litter		Soil		Litter		Soil		Soil	
	Spring	Autumn	Spring	Autumn	Spring	Autumn	Spring	Autumn	Spring	Autumn
Mesostigmata	0	13	1	9	0	3	0	4	0	3
Prostigmata	10	15	7	16	9	7	4	14	2	3
Cryptostigmata	19	20	8	30	2	5	1	10	0	0
Astigmata	1	1	1	1	1	1	1	1	1	1
Total	30	49	17	56	12	16	6	29	3	7
Total no. species recorded per habitat	64		67		25		32		10	

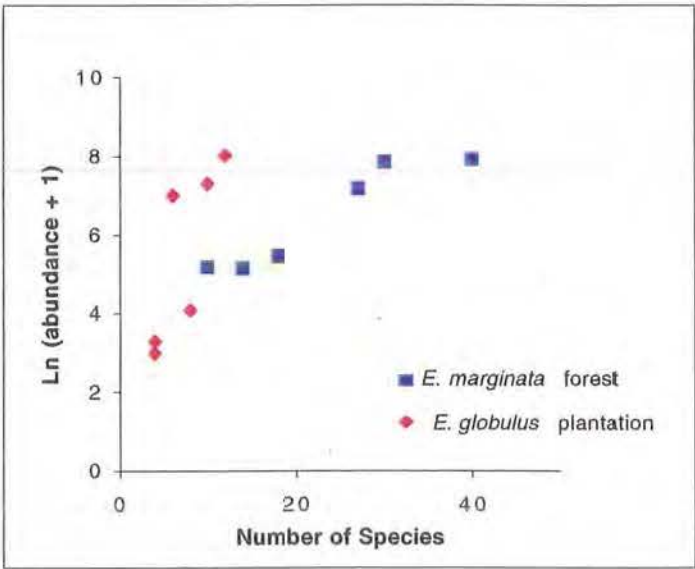


Figure 3.13. Log abundance-species richness of mites in surface litter of a native *E. marginata* forest and three 8 year-old *E. globulus* plantations.

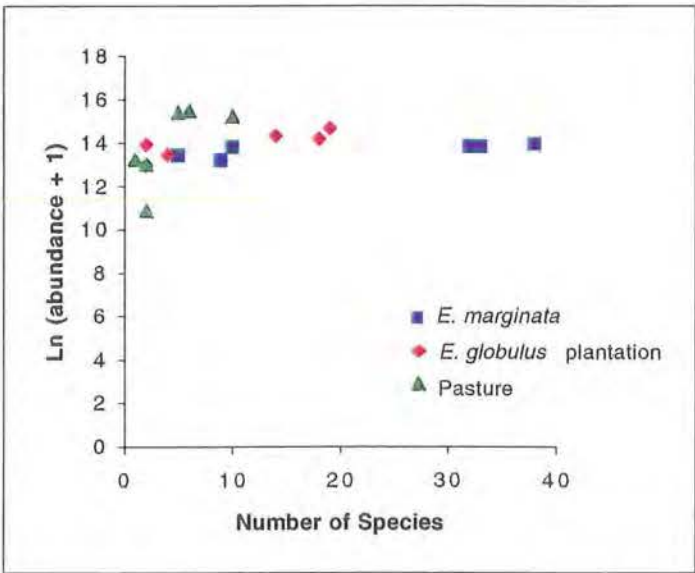


Figure 3.14. Log abundance-species richness plots of mites in soil under a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and a grazed pasture.

The increase in the number of species recorded between seasons is especially marked in the autumn soil samples of the native forest and *E. globulus* plantation. Those mite species extracted only from the autumn samples included the oribatids Tectocepheidae (*Tectocepheus velatus*, *Tegeozetes* sp.1), Oppiidae (sp.1, sp.2, sp.3, sp.4), Oppiinae (sp.1, sp.2, sp.3), Oppiellinae sp.1, Ceratozetidae sp.1, Mycobatidae sp.1, Oribatulidae (sp.2, sp.4), Brachychthoniidae (sp.1, sp.3), and the prostigmatids Pseudocheylidae (sp.3, sp.4), *Anoplocheylus* sp.1, Caeculidae sp.1, Eupodidae (sp.4, sp.5) and Rhagidiidae (sp.2, sp.4, sp.5).

Species richness was highest in the native forest in the soil and litter, being at least twice and sometimes three times that of the plantation and pasture sites. Species recorded only in the native forest were mainly Oribatida and several Prostigmata. The oribatids included *Haplochthoniuss* sp.2, *Tectocepheus* sp.2, *Eremaezetes* sp.1, *Scapheremaeus* sp.1, *Antarctiozetes* sp.1, *Mycobates* sp.1, *Scutovertex* sp.1, Plateremaeidae sp.1, Ceratozetidae sp.1, Phenopelopidae sp.1, Oribatulidae (sp.2, sp.4), Scheloribatidae (sp.1, sp.2), *Scheloribates* (sp.1, sp.2), and the prostigmatids Smarididae (sp.1, sp.2), Nanorchestidae sp.1, and *Linotetranus* sp.1.

Overall the soil and litter samples yielded roughly equivalent numbers of species within the native forest and *E. globulus* plantation. The pasture soil had very low species richness in comparison to the two forest ecosystems.

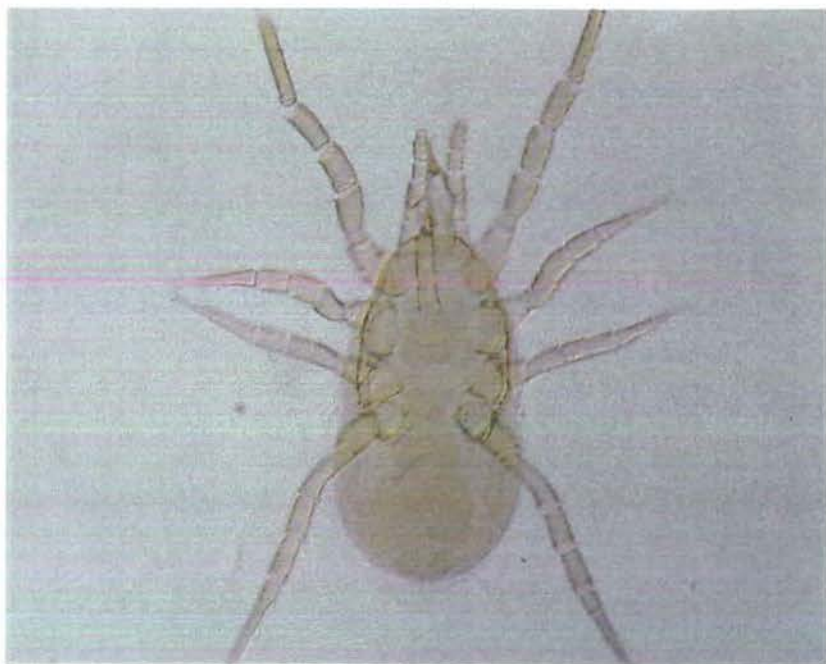


Plate 3.5a. Mesostigmata: Phytoseiidae sp.1, a member of the dominant family of predatory mites in most terrestrial ecosystems. Ventral view x 100.

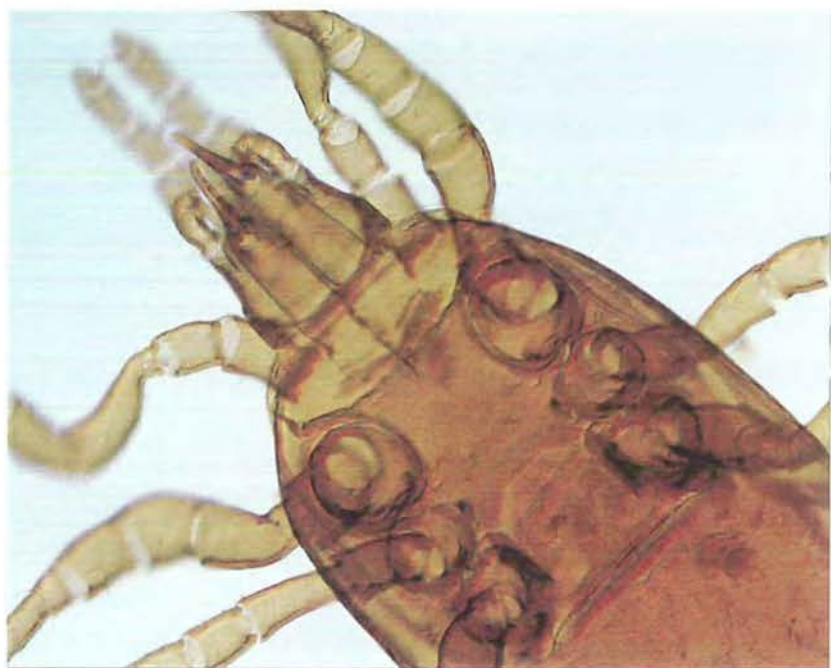


Plate 3.5b. Mesostigmata: Ologamasidae sp.1. Ventral view x 100.

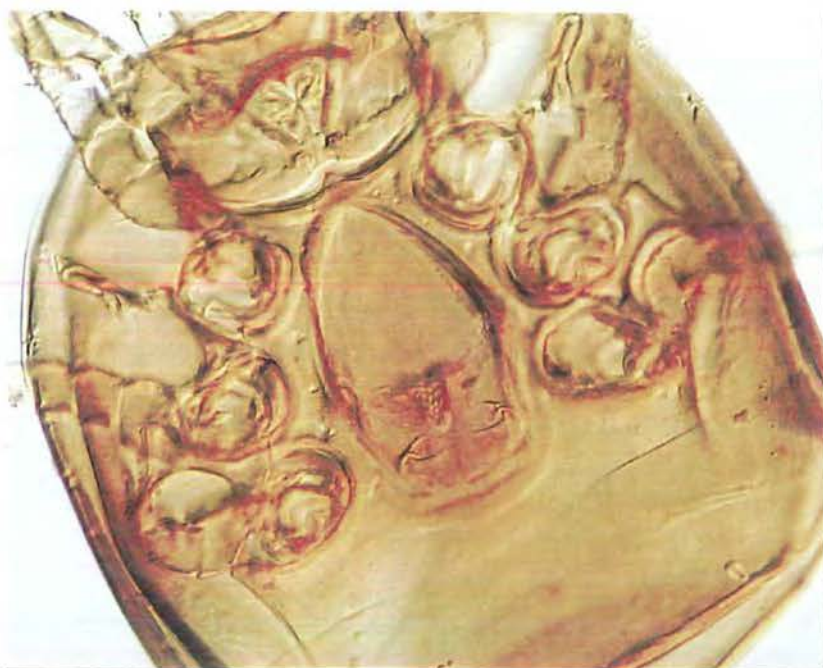


Plate 3.5c. Mesostigmata: Uropodidae sp.1. Ventral view x 400.

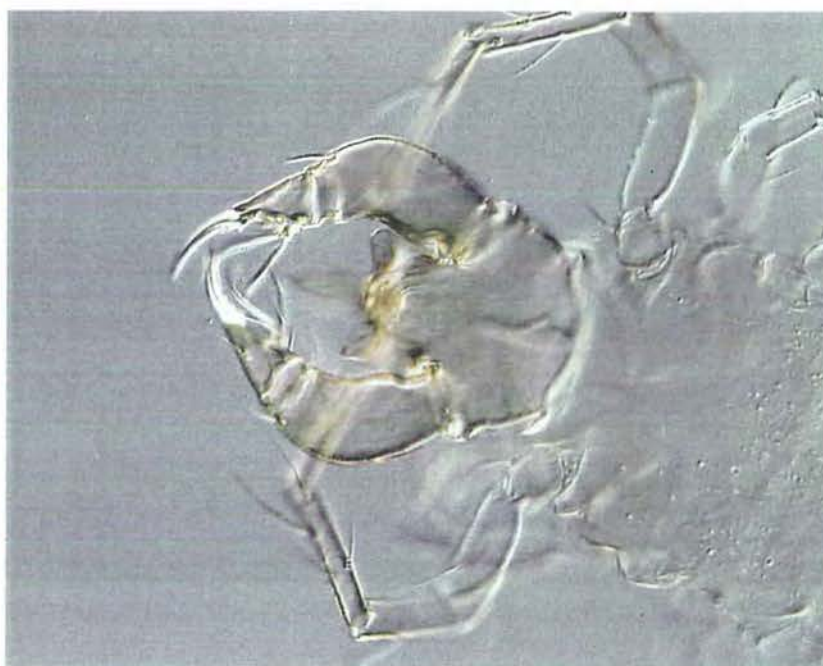


Plate 3.6a. Prostigmata: Cheyletidae sp.1 with large ornamented chelicerae typical of predaceous mites. Dorsal view x 400.

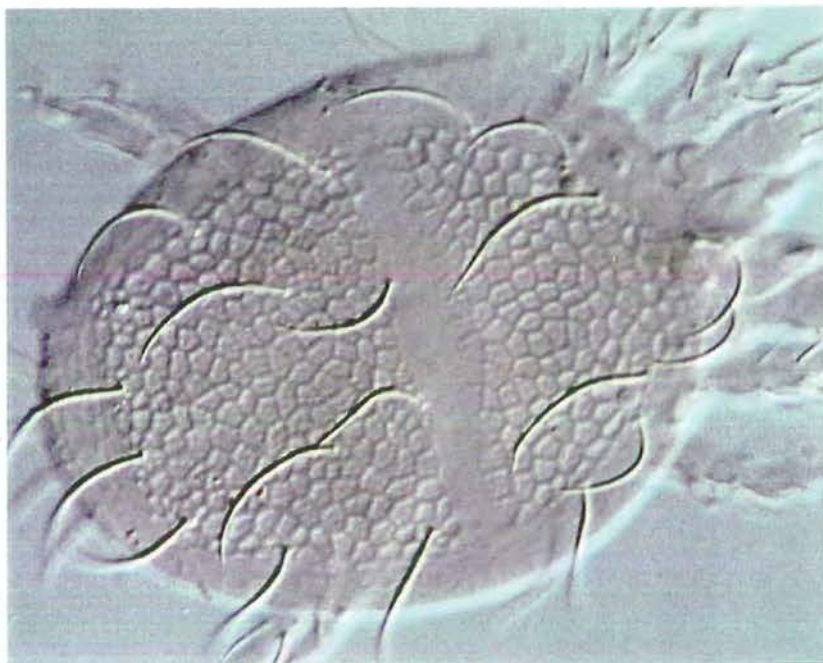


Plate 3.6b. Prostigmata: Stigmaeidae sp.1 with sclerotized body and long dorsal setae. Dorsal view x 400.

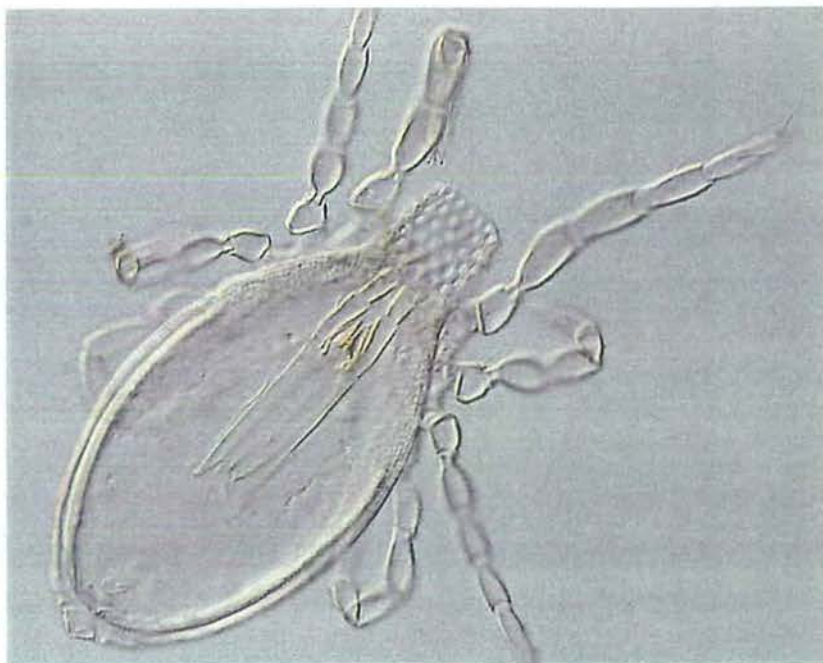


Plate 3.6c. Prostigmata: Cryptognathidae sp.1. Note the retractile gnathosoma in the idiosoma. Dorsal view x 400.



Plate 3.7a. Oribatida: Brachychthoniidae sp.1, a tiny typical lower oribatid mite.
Dorsal view x 1600.



Plate 3.7b. Oribatida: Oppiellinae sp.1. Ventral view x 400.

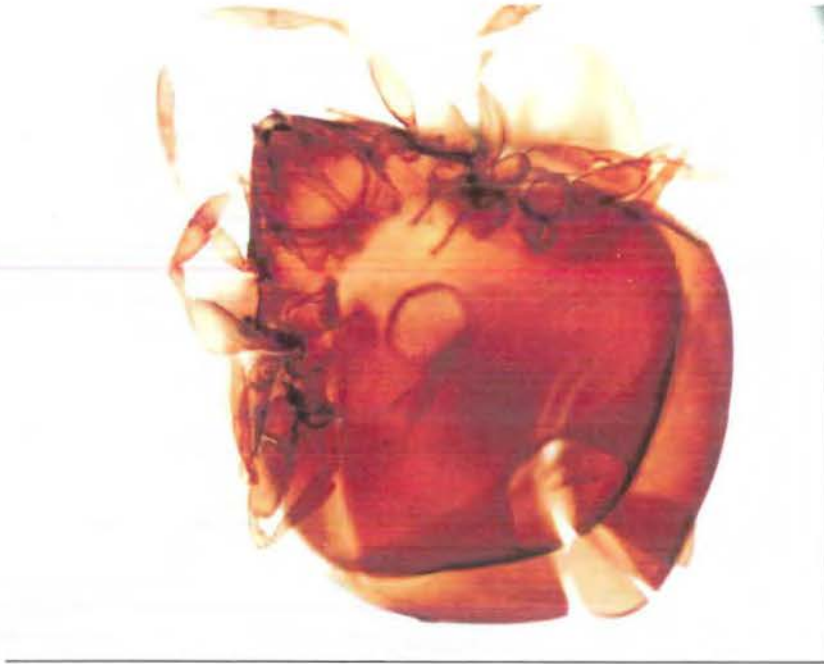


Plate 3.7c. Oribatida: Humerobatidae *Humerobates* sp.1, a higher oribatid mite.
Ventral view x 100.



Plate 3.7d. Oribatida: Scutoverticidae *Scutovertex* sp.1. Note the ornamented, sclerotized cerotegument. Ventral plate x 400.

3.5.2 Acarine Species Diversity

Diversity indices estimate the importance of species in the community, incorporating species richness and evenness (commonness and rarity) of species. The simplest measure of the character of a community is the non-parametric Simpson's reciprocal index of diversity ($1/D$) and is recommended by Krebs (1989) for studies that wish to weight the common species more. In this reciprocal form, Simpson's diversity can be most easily interpreted as the number of equally common species required to generate the observed heterogeneity of the sample, and makes no assumption about the shape of species abundance curves (Krebs, 1989; Begon *et al.*, 1996).

Since Simpson's diversity is calculated by determining, for each species, the proportion of individuals that it contributes to the total in the sample, those species with very high abundances (proportions) will obscure those species with lower abundances. Thus, there were several limitations to the usefulness of this index in the present study. Considering the exceptionally high abundances of the Astigmata, they were excluded from analyses due to their dominance masking the contribution of the other common, although less abundant species. Secondly, with the autumn samples having higher abundances and greater species richness than the spring samples, the sites sampled in autumn showed better community representation.

Species diversity of the surface litter in spring was higher in the *E. globulus* plantation (Table 3.12). Although species richness was considerably lower than the native forest, there was greater species evenness. Diversity in the autumn litter was considerably greater in the

native forest than the *E. globulus* plantation, due to both greater species richness (almost 3 times as many species) and higher species evenness. The lower abundances recorded in spring may have skewed the resultant species diversity, hence the pooling of the spring and autumn samples provided a more relevant outcome. Species diversity of the pooled seasons was substantially greater in the native forest due to the greater species richness and evenness.

Table 3.12. Community attributes of non-astigmatid species in surface litter under a native *E. marginata* forest and three 8 year-old *E. globulus* plantations.

Sample period	Site	Species Diversity (1/D)	Simpson's Evenness	Species Richness
Spring 1997	<i>E. marginata</i> forest	4.83	0.82	29
	<i>E. globulus</i> plantation	7.25	0.95	11
Autumn 1998	<i>E. marginata</i> forest	9.22	0.91	48
	<i>E. globulus</i> plantation	2.23	0.59	15
Total pooled seasons	<i>E. marginata</i> forest	12.44	0.94	63
	<i>E. globulus</i> plantation	2.51	0.63	24

The higher species richness of the native forest in the spring soil contributed to the greater diversity (Table 3.13). The very low number of species extracted from the plantation and pasture limited the usefulness of the diversity indices, particularly in the pasture. Autumn species diversity was slightly higher in the plantation than the native forest, yet both had similar species evenness. Species richness, however, was considerably higher in the native forest; almost double that of the plantation. Pooling of the seasons followed the trend

observed for the autumn sampling season, where diversity was slightly higher in the plantation compared to the native forest and was very low in the pasture.

Table 3.13. Community attributes of non-astigmatid species in soil of a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and a grazed pasture.

Sample period	Site	Species Diversity (1/D)	Simpson's Evenness	Species Richness
Spring 1997	<i>E. marginata</i> forest	7.62	0.93	16
	<i>E. globulus</i> plantation	4.50	0.93	5
	Pasture	-	-	2
Autumn 1998	<i>E. marginata</i> forest	2.55	0.62	55
	<i>E. globulus</i> plantation	3.03	0.69	28
	Pasture	1.16	0.15	6
Total pooled seasons	<i>E. marginata</i> forest	2.80	0.65	66
	<i>E. globulus</i> plantation	3.05	0.69	31
	Pasture	1.17	0.16	9

Clearly, species richness influences the degree of species diversity in the surface litter and soil. The greater richness in the litter layers of the native forest may reflect a more diverse environment.

3.5.3 Acarine Similarity Analysis

Similarity indices are useful measures of the extent to which two habitats have species in common, or have niche overlap (Southwood, 1987; Krebs, 1989). Morisita's index of similarity, as recommended by Krebs (1989), is the best measure of ecological similarity, as it is nearly independent of sample size, except for very small samples. It is best interpreted as the probability that an individual drawn from two distinct samples will belong to the same species and the probability that two individuals drawn from either sample will belong to the same species (Krebs, 1989).

As with the species diversity indices, there were several limitations to the usefulness of the similarity indices. Morisita's index is based on counts of individuals. If one species has exceptionally high numbers, the probability of that species being represented increases substantially, such that the probability of the other species being represented is concealed. Again, the Astigmata were excluded from analyses due to their very high numbers, with their incorporation into the analyses masking potentially important differences in the less abundant species. Secondly, the very poor abundance and richness of the spring samples provided very little useful information. Hence, the pooling of seasons permitted a better interpretation of Morisita's index.

Species composition in the surface litter of the *E. globulus* plantation (Morisita's index = 0.49) was half that of the native forest (Morisita's index = 1.00). From a total of 69 non-astigmatid species recorded in the litter, 17 species were extracted from both systems. These were 3 Mesostigmata (Phytoseiidae *sp.1*, Ologomasidae *sp.2* and Ascidae *sp.1*), 8

Prostigmata (*Anoplocheylus sp.1*, *Odontoscirus sp.1* and *sp.3*, *Spinibdella sp.1*, Cheyletidae *sp.1*, Stigmaeidae *sp.1*, Tarsonemidae *sp.1* and Tetranychidae *sp.1*) and 6 Oribatida (*Tectocephus velatus*, *Tegeozetes sp.1*, Oppiinae *sp.1* and *sp.3*, Oppiellinae *sp.1* and Mycobatiidae *sp.1*).

Species composition in the soil was more similar between the native forest and pasture than between the native forest and plantation (Table 3.14). The high similarity index between the native forest and pasture (0.90) is because all the species extracted from the pasture sites were found in the native forest. Thus, the pasture was more similar to the native forest rather than the native forest being more similar to the pasture. More species were shared between the forest and plantation, with the majority being in low abundance.

Table 3.14. Morisita's index of similarity matrix of non-astigmatid species composition in soil of a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and a grazed pasture.

	Site	Forest	Plantation	Pasture
Pooled seasons	<i>E. marginata</i> forest	1.00		
	<i>E. globulus</i> plantation	0.75	1.00	
	Pasture	0.90	0.66	1.00

Excluding the Astigmata, the pasture and native forest shared 6 species from a total of 82 species extracted from the soil; 3 Mesostigmata (Phytoseiidae *sp.1*, Dermanyssoidea *sp.1* and Unknown *sp.1*) and 3 Prostigmata (*Odontoscirus sp.1*, Penthalodidae *sp.2* and Tetranychidae *sp.2*). Sixteen species from the total of 82 were recorded in the soil under both the native forest and plantation. These were 3 Mesostigmata (Phytoseiidae *sp.1*,

Ascidae *sp.1* and Dermanyssoidea *sp.1*), 3 Prostigmata (*Odontoscirus sp.2*, *Spinibdella sp.1* and Raphignathidae *sp.2*) and 10 Oribatida (*Haplochthonius sp.1*, *Tegeozetes sp.1*, Oppiinae *sp.1*, *sp.2* and *sp.3*, Oppiidae *sp.1* and *sp.2*, Oppiellinae *sp.1*, Mycobatiidae *sp.1* and Unknown *sp.1*). The plantation and pasture recorded 3 species in common, 2 Mesostigmata (Phytoseiidae *sp.1* and Dermanyssoidea *sp.1*) and 1 Prostigmata (Rhagidiidae *sp.5*).

Several species were extracted from both the soil and litter layers in both communities; 2 Mesostigmata (Phytoseiidae *sp.1* and Ascidae *sp.1*), 1 Prostigmata (*Spinibdella sp.1*) and 5 Oribatida (Oppiinae *sp.1* and *sp.3*, Oppiellinae *sp.1*, *Tegeozetes sp.1* and Mycobatiidae *sp.1*).

3.5.4 Rank Abundance (Whittaker) Curves

Species abundance relations (the relations between abundance and the number of species possessing that abundance) can be illustrated utilizing log abundance-species ranks (May, 1975) (Figures 3.13 and 3.14). These plots emphasize that the relationship between abundance of individuals and species numbers has two features: species richness (the total number of species present in the area) and equitability or evenness (the pattern of distribution of individuals between the species) (Southwood, 1987). The equitability of the relationship will be a reflection of the underlying distribution or organization of the (mite) community and is illustrated by the shape of the resultant curve (May, 1975; Southwood, 1987; Krebs, 1989).

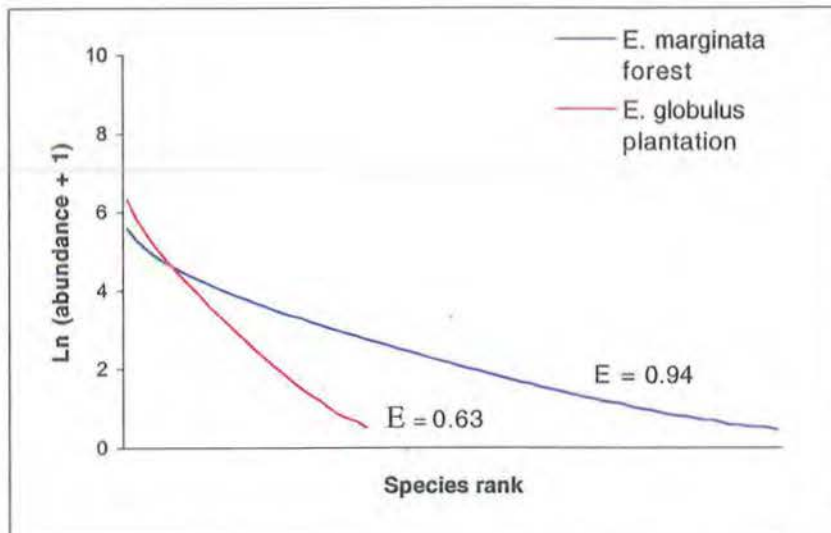


Figure 3.13. Whittaker plots of all species in surface litter of a native *E. marginata* forest and three 8 year-old *E. globulus* plantations. (E = Simpson's evenness).

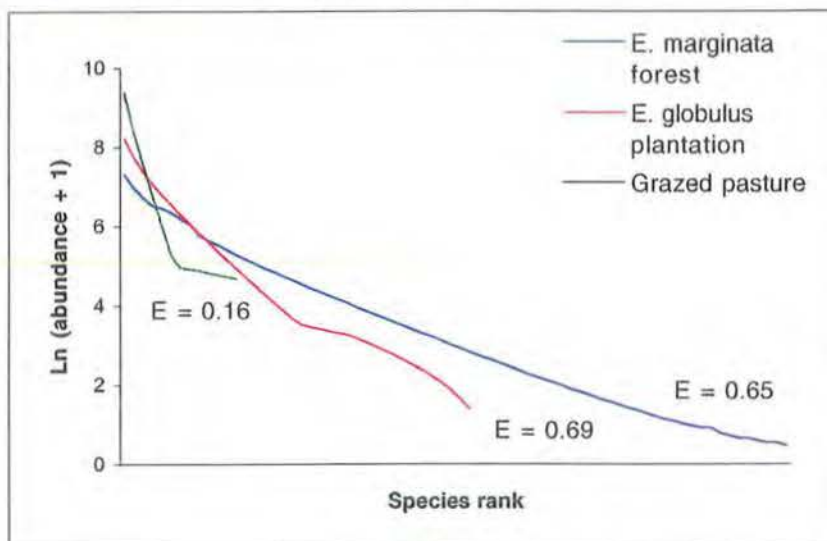


Figure 3.14. Whittaker plots of all species in soil under a native *E. marginata* forest, three 8 year-old *E. globulus* plantations and a grazed pasture. (E = Simpson's evenness).

The surface litters of the native forest and *E. globulus* plantation had similar trends in their patterns of species distributions (Figure 3.13). The curve of the *E. globulus* plantation illustrates a geometric series distribution (as the ideal case) and the native forest illustrates a log series distribution (as the statistically realistic expression of the geometric series) (May, 1975). These distributions illustrate the division of the niche volume proceeding in a strong hierarchical fashion, such that both litter communities contain comparatively few species that are common (in higher abundances) and comparatively large numbers of species that are rare (in lower abundances) (May, 1975; Krebs, 1989). This distribution also illustrates that abundances vary more on a seasonal basis. The log series represents the arrival of species at random time intervals, and could be influenced by the broader chemistry of the associated heterogeneous litter. The geometric series niche-preemption mechanism stems from the fact that species arrive at successive uniform time intervals and proceed to intercept (preempt) a fraction of the remaining niche before the arrival of the next species (May, 1975). Since litter input is continuous and has a monospecific nature, the chemical constituents would be expected to decompose at uniform rates.

The soil communities had considerable variation in the species-abundance relationship between the three ecosystems (Figure 3.14). As with the litter, the soil mites of the native forest had a log series distribution. The soil community under the *E. globulus* plantation varied in distribution to the litter community, showing a broken-stick distribution. This distribution is expected whenever an ecologically homogeneous group of species apportion randomly among themselves a fixed amount of a major environmental resource (May, 1975), and in this instance, may well be due to the monospecific litter decomposing at uniform times. Interpretation of the pasture rank-abundance plot is limited by the shortfall

of the distribution curve, due to the resultant low number of species extracted from the ecosystem. Considering this, the curve appears as a lognormal distribution. Of the two different ecological regimes associated with a lognormal distribution, the opportunistic regime is most likely to fit the pasture ecosystem, which is characterized by predominant random environmental fluctuations determining the relative abundances of populations (May, 1975). In contrast, the equilibrium regime is associated with random variables and communities comprising a large number of species fulfilling diverse roles, with interactions within the community controlling all populations around steady values (May, 1975).

Lognormal distributions occur in environments associated with products of random variables or when several factors become significant, such as influxes in the resource base.

Southwood (1987) concludes from theoretical studies (May, 1975) that the rank-abundance models can be arranged in a series corresponding to maximal niche-preemption or unevenness and moving through to a more uniform resource apportionment. Thus:

(uneven) geometric series → log series → log normal → broken-stick (even). Southwood's conclusion is demonstrated in the native forest and *E. globulus* litter communities of the present study, where the log series distribution of the native forest corresponded with greater evenness than the more uneven geometric distribution of the *E. globulus* plantation.

In the soil, the broken-stick distribution of the *E. globulus* plantation was slightly more even than the log series of the native forest, also confirming Southwood's (1987) conclusion. The lognormal distribution of the pasture would be expected to have a greater evenness than the log series (native forest), yet was considerably lower due to the very low number of species extracted from the ecosystem.

3.5.5 Multivariate Analyses of Acari within the Surface Litter

Ordination

Multi-dimensional plots were produced, with a three-dimensional plot providing adequate representation of species assemblages in sites within the surface litter (Figure 3.15). The three-dimensional plot had an acceptable stress value of 0.10, corresponding to a good ordination with no real risk of drawing false inferences (Clarke, 1993). Discrimination of sites was best illustrated on the ordination axes 1 and 2 in the two-dimensional plot.

There was distinct separation of the samples by season, with spring separating from autumn along axis 2. This would be expected, considering the lower abundances and species numbers extracted from the spring samples. There are also large ordination distances between the spring native forest sites and autumn native forest sites 1 and 2, reflecting the differences in species richness. The little separation of the spring plantation cluster from the autumn plantation sites along the axes suggests similar species assemblages between sites, which is not unexpected given the very low numbers of species abundances.

There is clear separation of the spring native forest sites from the spring plantation sites, suggestive of different species assemblages. There was very little overlap of species between the native forest and plantation sites in spring, with only 3 species recorded in both (Prostigmata: *Odontoscirus sp.1* and *Stigmaeidae sp.1*, and Astigmata: *Tyrophagus sp.1*).

The three spring plantation sites are positioned close together, with sites

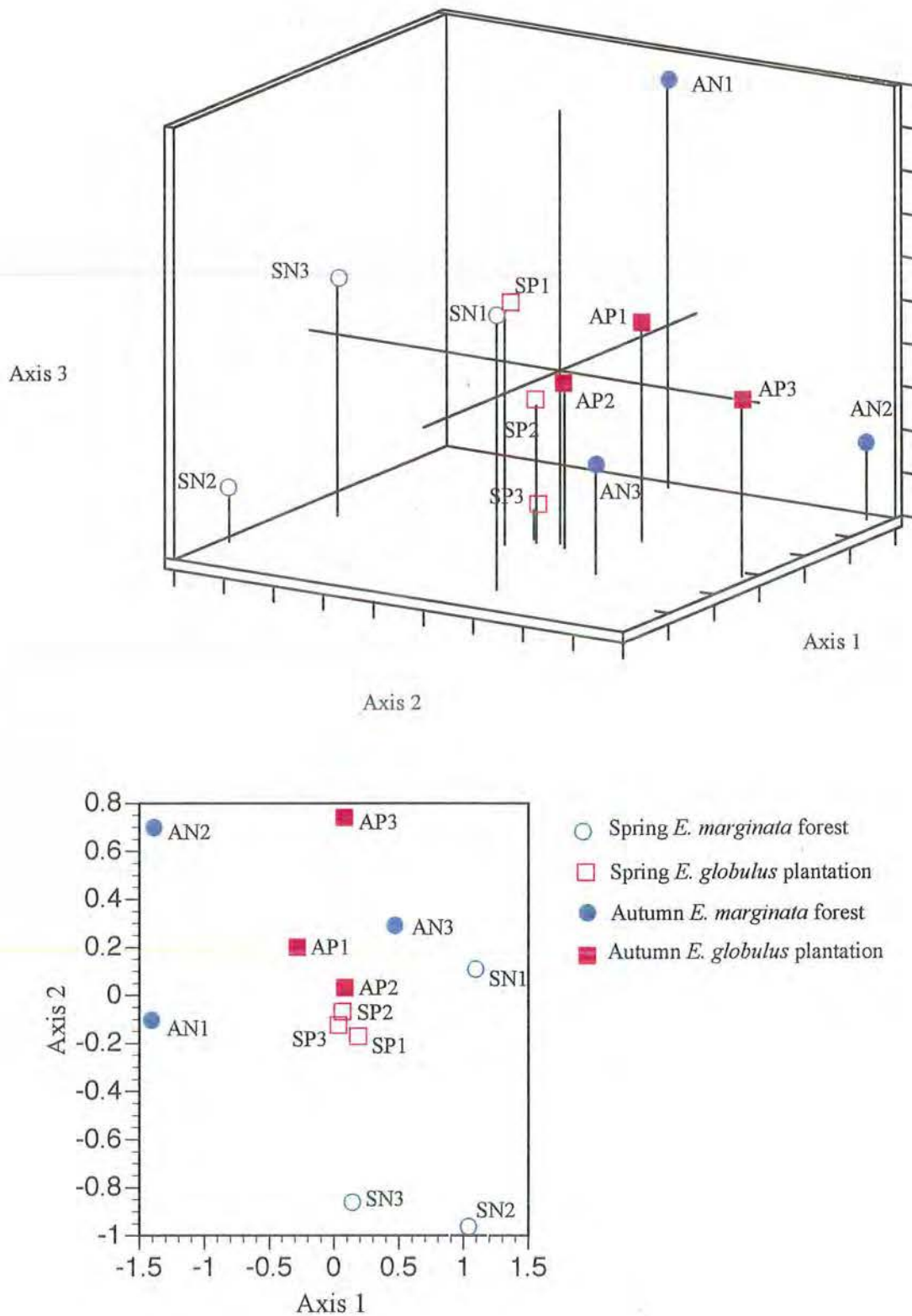


Figure 3.15. SSH ordination in three and two dimensions of acarine communities in surface litter of a native *E. marginata* forest and an 8 year-old *E. globulus* plantation. Stress = 0.10.

2 and 3 grouping closer. This could be reflecting the presence of two species, *Odontoscirus sp.1* in low abundance and *Tyrophagus sp.1* in very high abundance. Only 1 species, *Stigmaeidae sp.1*, was present in low abundance at all plantation sites. In the spring native forest sites, species assemblages within each site differed greatly from one other, with only 1 species (Cryptostigmata: *Mycobates sp.1*) recorded in all sites, and in very high abundance. Native forest sites 1 and 2 aligned similarly along axis 1, recording 3 species in common, *Smarididae sp.2* (Prostigmata) and *Oribatulidae sp.4* (Cryptostigmata) in low abundances and *Scapheremaeus sp.1* (Cryptostigmata) in very high abundance. Native forest sites 2 and 3 aligned similarly along axis 2, recording 3 species in common and in similar low abundances (Prostigmata: *Rhagidiidae sp.1* and *Stigmaeidae sp.1*, and Cryptostigmata: *Humerobates sp.1*).

In autumn there is clear separation of native forest sites 1 and 2 along axis 1. Fourteen species were recorded at both sites, 1 Mesostigmata (*Phytoseiidae sp.1*), 1 Prostigmata (*Anoplocheylus sp.1*) and 1 oribatid (*Oppiinae sp.1*) were in very high abundances, whilst the oribatid *Ceratozetidae sp.1* and 7 Mesostigmata (*Ologomasidae sp.2*, *Ologomasidae sp.3*, *Eviphididae sp.1*, *Ascidae sp.1*, *Dermanyssoidea sp.1*, *Unknown sp.1* and *Unknown sp.2*) were all recorded in low abundances. Site 3 had the lowest species richness among the native forest sites, positioning within the cluster of plantation sites with their lower species richness and diversity.

Classification

The pattern of mites within the surface litter is relatively robust when the ordination outcomes along axis 1 are compared with the hierarchical agglomerative pattern (Figure 3.16) The first group distinguished is the autumn native forest sites 1 and 2, clustering at a late stage in the dendrogram with the greatest dissimilarity in species assemblage from the other sites. The next interpretable cluster of sites is the largest and includes all the spring and autumn plantation sites, of which the spring sites group separately to the autumn sites. Once again, autumn native forest site 3 clusters among the autumn plantation sites. The final group of sites to be distinguished at an early stage in the dendrogram is the cluster of the three spring native forest sites. The large dissimilarity distance between the first and final groups reflects the lower species richness and lower abundances recorded in the spring sampling season.

Summary

Within the litter habitat, these classification and ordination analyses are suggestive of distinct species assemblages between native forest and plantation sites. Seasonal effects enhance these differences, reflecting the greater number of mites and greater species richness extracted from the autumn samples, thus providing better data sets for analyses.

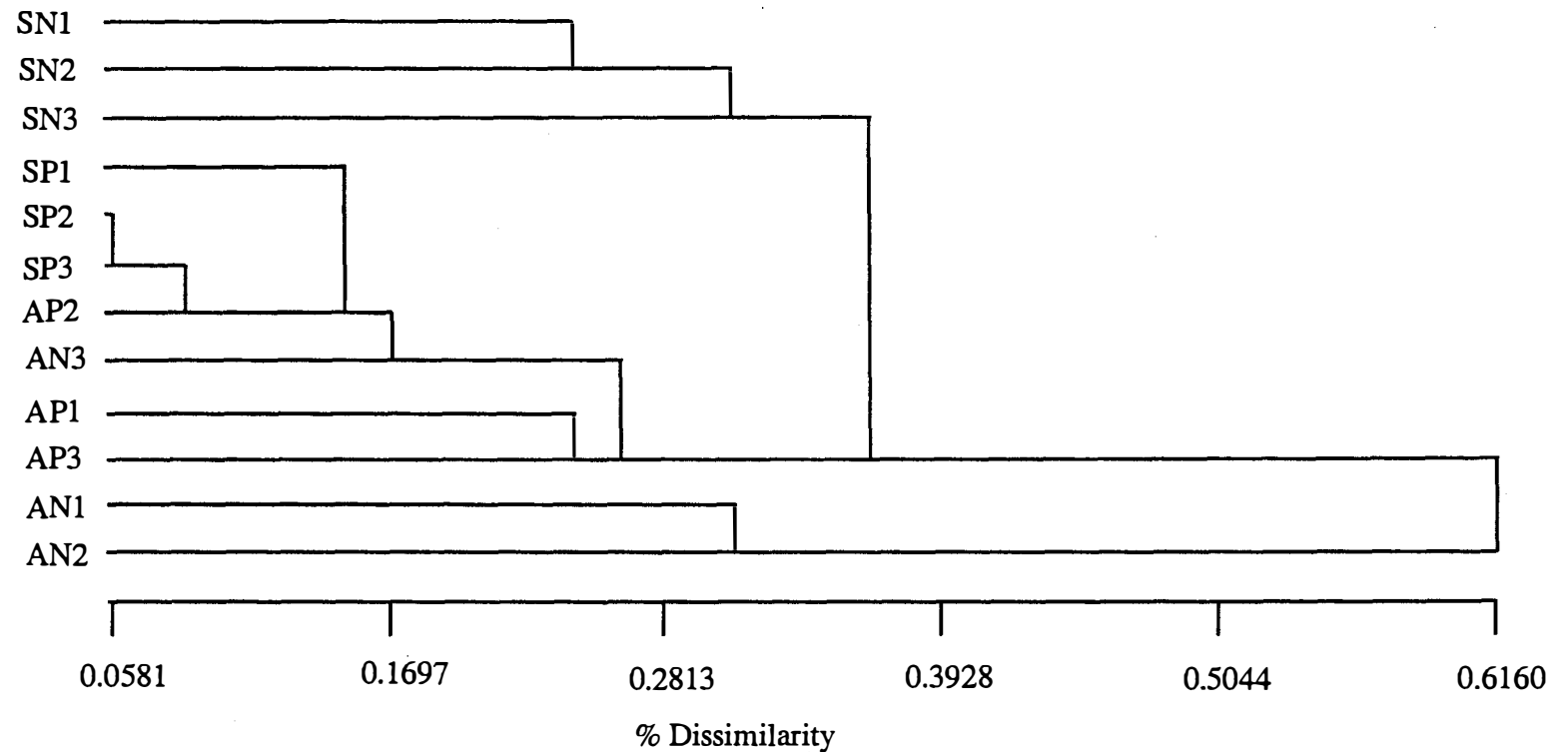


Figure 3.16 Hierarchical agglomerative clustering of surface litter acarine species under an 8 year-old *E. globulus* plantation and native *E. marginata* forest. (SN1, SN2, SN3: spring native forest sites; SP1, SP2, SP3: spring plantation sites; AN1, AN2, AN3: autumn native forest sites; AP1, AP2, AP3: autumn plantation sites).

3.5.6 Multivariate Analyses of the Soil Acari

Ordination

As with the SSH ordination of the surface litter, the three-dimensional plot of the soil produced an acceptable stress value of 0.16 (Clarke, 1993) (Figure 3.17). In the two-dimensional plot discrimination of sites was best illustrated on the ordination axes 1 and 2 (Figure 3.17). Again, as for the litter, seasonal effects on the species assemblages in the soil were most obvious, with clear separation of the spring sites from the autumn sites along axis 1. However, it is important to note that the low numbers limits the usefulness of the ordination procedure.

In both seasons there was separation of the native forest sites from the cluster of plantation and pasture ecosystems. In spring, the native forest sites separated from the plantation and pasture sites along axis 1. Considering the very low number of species recorded in the spring native forest, only 2 species were recorded at all 3 sites (Cryptostigmata: *Mycobates sp.1* and Astigmata: *Tyrophagus sp.1*). The autumn native forest sites distinctly separated from the autumn plantation and pasture sites along axis 2. The three native forest sites shared 11 species, including the Astigmatid *Tyrophagus sp.1* in very high abundance. Three Mesostigmata (Phytoseiidae *sp.1*, Ascidae *sp.1* and Unknown *sp.1*) and 2 Prostigmata (*Anoplocheylus sp.1* and Tarsonemidae *sp.1*) were in similar low abundances. *Odontoscirus sp.1* (Prostigmata) was highly abundant at native forest site 1, Mesostigmata Unknown *sp.1* very high at site 2 and *Spinibdella sp.1* (Prostigmata) exceptionally abundant at site 3.

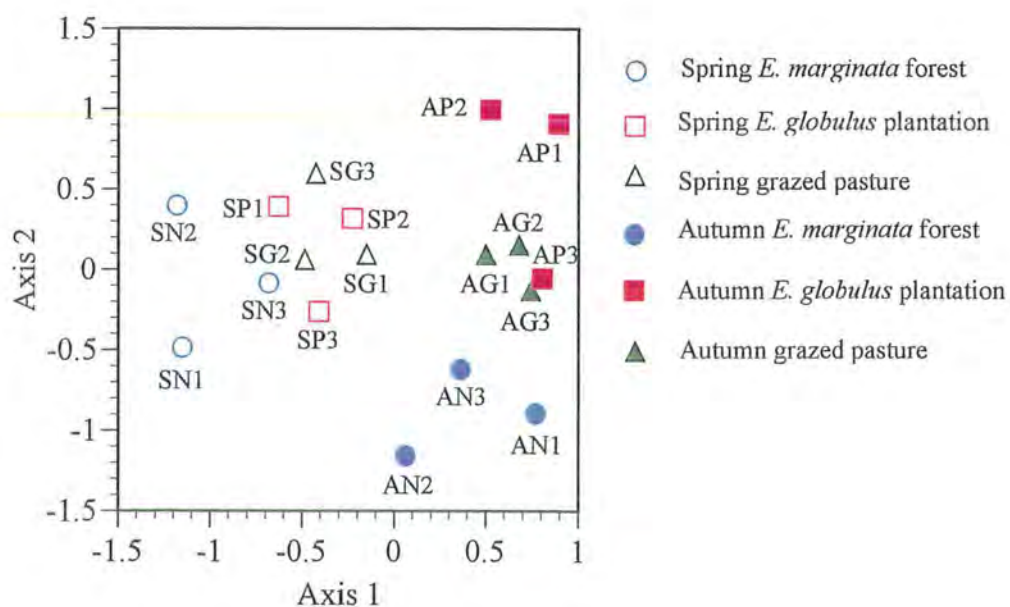
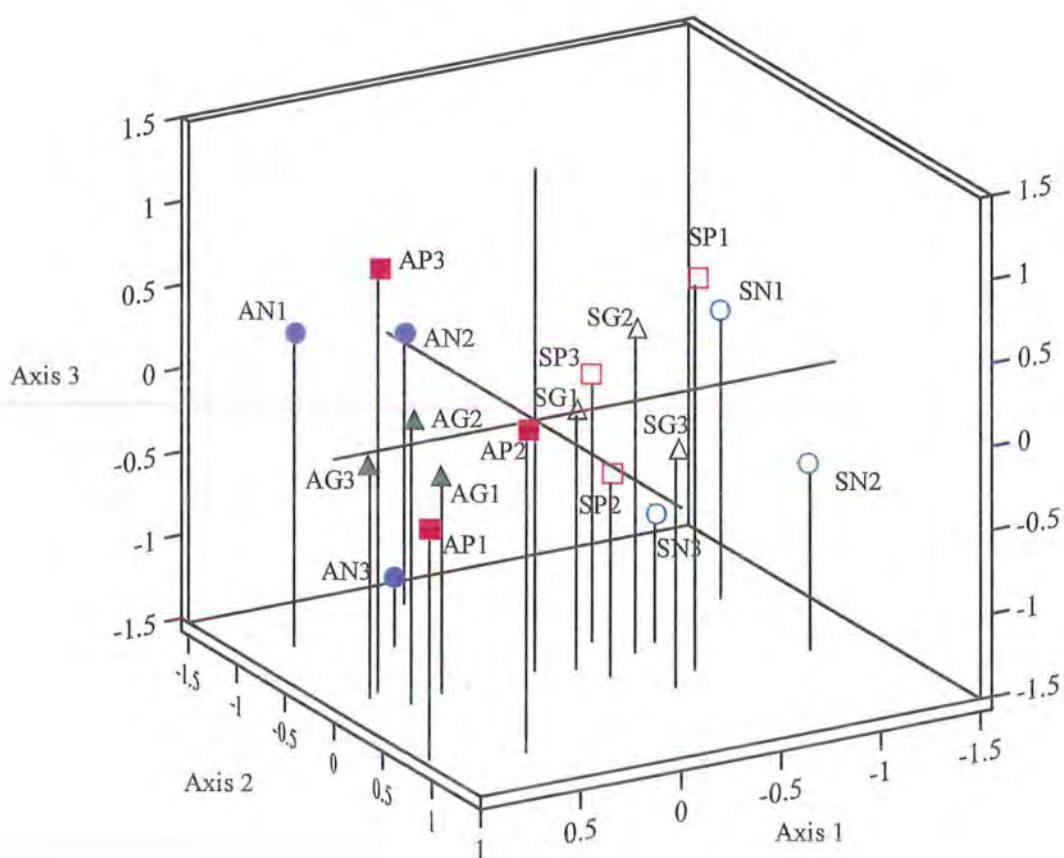


Figure 3.17. SSH ordination in three and two dimensions of acarine communities in soil under a native *E. marginata* forest, an 8 year-old *E. globulus* plantation and grazed pasture. Stress = 0.16.

The lower oribatid *Haplochthonius sp.2* was very abundant at sites 1 and 3, as was the oribatid, Oppiidae *sp.1* at sites 1 and 2.

In addition to being separated from the native forest sites, both the plantation and pasture sites clustered in both seasons. In spring there was no clear separation of the plantation and pasture sites, likely due to the very low species richness and shared absences of many species, such as the Mesostigmata. The autumn plantation sites 1 and 2 had some separation along axis 2 from autumn plantation site 3 and the autumn pasture sites, having similar high abundances of Phytoseiidae *sp.1* (Mesostigmata), Eupodoidea *sp.1* (Prostigmata), and the oribatid Oppiidae *sp.2*. Plantation site 1 also had the higher abundance of Oppiinae *sp.3* (Cryptostigmata) whilst site 2 had the higher abundance of Oppiellinae *sp.1* (Cryptostigmata). Ordination distances between the three autumn pasture sites and plantation site 3 were small, reflecting similar species assemblages. These pasture sites also had very high abundances of a few species (Astigmata: *Tyrophagus sp.1*, Mesostigmata: Phytoseiidae *sp.1*) and very low abundances of the remaining species (Prostigmata: Cunaxidae *sp.3*, Penthalodidae *sp.2*, Rhagidiidae *sp.5*, Tetranychidae *sp.2* and Erythraeidae *sp.3*).

Classification

The hierarchical agglomerative clustering of the soil mites (Figure 3.18), revealed the same seasonal patterns, with two groups (spring and autumn sites) separating from each other.

The most apparent cluster includes the native forest sites, showing the greatest dissimilarity from the other ecosystems. Within the autumn group, the distance between the cluster of the autumn native forest sites and the plantation and pasture sites was present yet not greatly separated.

Summary

As with the litter assemblages, the soil mite species assemblages showed seasonal differences. The site differences between soil assemblages were not as obvious as in the litter. This is not unexpected, given that the impact of the differences in the litter habitat need not necessarily be translated to the soil habitat.

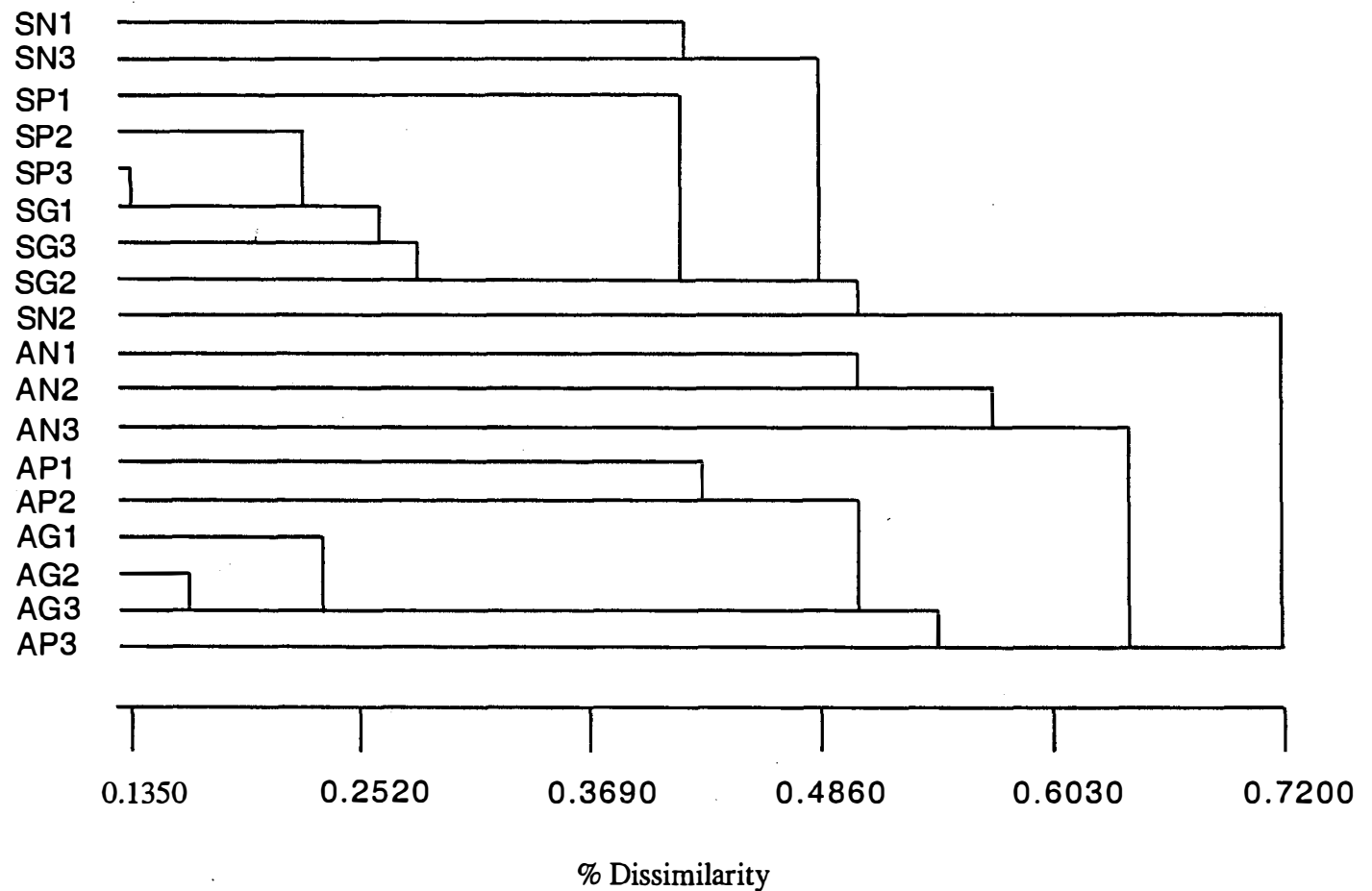


Figure 3.18. Hierarchical agglomerative clustering of soil acarine species in an 8 year-old *E. globulus* plantation, native *E. marginata* forest and grazed pasture. (SN1, SN2, SN3: spring native forest sites; SP1, SP2, SP3: spring plantation sites; SG1, SG2, SG3: spring pasture sites; AN1, AN2, AN3: autumn native forest sites; AP1, AP2, AP3: autumn plantation sites; AG1, AG2, AG3: autumn pasture sites).

3.6 Further Analysis of the Cryptostigmata (Oribatida)

Oribatid mites are considered by several authors (Wallwork, 1967; Seastedt, 1984; Coleman and Crossley, 1996; Franchini and Rockett, 1996; Behan-Pelletier, 1999) to be of ecological significance as bioindicators of soil quality and fertility at the species level. For this reason, the distribution of the oribatid species in the sampled systems is described here in detail.

3.6.1 Species Richness

A total of 45 oribatid mite species was recognized from the soil-litter system, representing 19 recognized families. Of these, 34 species were recorded only in the native forest and 12 in both ecosystems, with no species exclusively recorded under the *E. globulus* plantation (Table 3.15). No oribatid mites were extracted from the grazed pasture samples.

Of those species recorded only in the native forest, 4 species were recorded in relatively high abundances in the surface litter and soil, Brachychthoniidae *sp.3* (112 kg⁻¹ dry litter weight and 816 m⁻² soil), Brachychthoniidae *sp.1* (38 kg⁻¹ and 1071 m⁻²), *Mycobates sp.1* (57 kg⁻¹ and 357 m⁻²), and *Scapheremaeus sp.1* (37 kg⁻¹ and 51 m⁻²), and *Haplochthonius sp.2* (1531 m⁻²) in the soil. The remaining species were all recorded in very low abundances both in the litter (< 10 kg⁻¹) and soil (< 204 m⁻²).

Table 3.15. Presence (+) and absence (-) of oribatid mite species in the soil-litter subsystem of a native *E. marginata* forest and three 8 year-old *E. globulus* plantations.

Species	<i>E. marginata</i> forest	<i>E. globulus</i> plantation
Haplochthoniidae		
<i>Haplochthonius sp.1</i>	+	+
Mycobatiidae		
<i>Sp.1</i>	+	+
Oppiidae		
<i>Sp.1</i>	+	+
<i>Sp.2</i>	+	+
Oppiinae		
<i>Sp.1</i>	+	+
<i>Sp.2</i>	+	+
<i>Sp.3</i>	+	+
Oppiellinae		
<i>Sp.1</i>	+	+
Oribatulidae		
<i>Sp.6</i>	+	+
Tectocephidae		
<i>Tectocephus velatus</i>	+	+
<i>Tegeozetes sp.1</i>	+	+
Unknown <i>sp.1</i>	+	+
Astegistidae		
<i>Cultroribella sp.1</i>	+	-
Brachychthoniidae		
<i>Sp.1</i>	+	-
<i>Sp.3</i>	+	-
Ceratozetidae		
<i>Sp.1</i>	+	-
Cymbaeremaeoidea		
<i>Scapheremaeus sp.1</i>	+	-
Damaeidae		
<i>Sp.1</i>	+	-
Eremaozetidae		
<i>Eremaozetes sp.1</i>	+	-
Galumnidae		
<i>Sp.1</i>	+	-
Haplochthoniidae		
<i>Haplochthonius sp.2</i>	+	-
Haplozetidae		
<i>Sp.1</i>	+	-
Humerobatidae		
<i>Humerobates sp.1</i>	+	-
Mycobatidae		
<i>Antarctiozetes sp.2</i>	+	-

Table 3.15. continued

Species	<i>E. marginata</i> forest	<i>E. globulus</i> plantation
<i>Mycobates sp.1</i>	+	-
Oppiidae		
<i>Sp.3</i>	+	-
<i>Sp.4</i>	+	-
<i>Sp.5</i>	+	-
<i>Sp.6</i>	+	-
Oribatulidae		
<i>Sp.1</i>	+	-
<i>Sp.2</i>	+	-
<i>Sp.4</i>	+	-
<i>Sp.5.</i>	+	-
Phenopelopidae		
<i>Sp.1</i>	+	-
Plateremaeidae		
<i>Sp.1</i>	+	-
Scutoverticidae		
<i>Scutovertex sp.1</i>	+	-
Scheloribatidae		
<i>Scheloribates sp.1</i>	+	-
<i>Scheloribates sp.2</i>	+	-
<i>Sp.1</i>	+	-
<i>Sp.2</i>	+	-
Spherochthoniidae		
<i>Sp.1</i>	+	-
Tectocephidae		
<i>Tectocephus sp.2</i>	+	-
<i>Sp.1</i>	+	-
Unknown <i>sp.2</i>	+	-
Unknown <i>sp.3</i>	+	-

Of the 12 species recorded in both the native forest and *E. globulus* plantation sites, 5 species had similar abundances in both soil communities, Oribatulidae *sp.6*, *Tegeozetes sp.1*, Oppiinae *sp.2*, Mycobatiidae *sp.1* and Unknown *sp.1*. Species with similar litter abundances included Oppiidae *sp.1* and *Tegeozetes sp.1*.

Species that had relatively higher abundances in the litter of the native forest compared to the *E. globulus* litter included Mycobatiidae *sp.1* (13 kg⁻¹), *T. velatus* (14 kg⁻¹) and

Oppiinae *sp.3* (77 kg⁻¹), and those in the soil included *Haplochthonius sp.1* (714 m⁻²), Oppiinae *sp.1* (561 m⁻²) and Oppiidae *sp.1* (1021 m⁻²). Species only present in the litter of the native forest were Oppiidae *sp.1* and Oppiinae *sp.3* in low abundances. Those species in relatively higher abundances in the soil under the *E. globulus* plantation included Oppiinae *sp.3* (6071 m⁻²) and Oppiidae *sp.2* (612 m⁻²). Oppiellinae *sp.1* was high in the litter (29 kg⁻¹) and exceptionally high in the soil (25,051 m⁻²) under the plantation.

3.6.2 Species Diversity

Oribatid species diversity was greater in both the soil and litter layers of the native forest (Table 3.16). This was due to both the higher species richness and higher species evenness. Species diversity in the native forest was particularly high in the soil habitat.

Table 3.16. Community attributes of oribatid mite species in surface litter and soil of a native *E. marginata* forest and three 8 year-old *E. globulus* plantations.

Pooled seasons	Site	Species Diversity (1/D)	Simpson's Evenness	Species Richness
Surface Litter	<i>E. marginata</i> forest	9.18	0.92	33
	<i>E. globulus</i> plantation	3.37	0.82	7
Soil	<i>E. marginata</i> forest	15.19	0.96	35
	<i>E. globulus</i> plantation	1.99	0.54	14

CHAPTER 4: DISCUSSION

The inclusion of soil organisms in the formulation and refinement of ecological theories is a relatively recent approach that aims to provide better understanding of the functioning of below-ground ecosystems (Ohtonen, Aikio and Vare, 1997). This is an important objective given that the overall functioning of terrestrial ecosystems is dependent upon the interactions and interrelationships between the aboveground and belowground food webs. Better understanding will aid sustainable management of natural and man-modified ecosystems. Currently, there are no specific ecological frameworks for the assessment of belowground ecosystems, with such studies based on frameworks developed from investigations of the ecosystems aboveground. Theoretical frameworks applied to the present study involve Hutchinson's niche theory (1957) and, to a lesser extent, trophic structure analysis. Trophic analyses are the dominant model used for assessing the health of the soil ecosystem, and will be used in this discussion to assess how the effect of resource limitation (bottom-up effects) affects the different trophic levels (Wardle and Giller, 1997).

The analysis of changes to the belowground ecosystem described in this present study can be approached from several perspectives. A habitat disturbance gradient can be perceived along the three different vegetation systems. The sequence begins with the highly simplified grazed pasture with no litter layer, to the monoculture plantation with its monospecific litter, and finally to the adjacent native jarrah forest with its floral complexity and diversity, soil-litter temperature, pH and heterogeneous litter layer. This same habitat disturbance gradient can represent succession and chronology. Long-lived communities of native jarrah forest have been totally cleared of all vegetation on a large scale and replaced with

pastureland. After many years of non-native livestock grazing, some of these pastures have been re-vegetated with monoculture plantations. This chronological sequence of events will be the framework for the following discussion.

The influence each disturbance has on the soil and litter mesofauna can be assessed in several ways. Initially, the assessment of the taxonomic diversity and species richness of the mite community will provide some indication of the ecological complexity of the different soil communities (Behan-Pelletier, 1999). Moreover, the effect of simplifying the litter and altering the resource niche can be assessed using functional trophic groups. Plant matter entering the soil represents the initial food source on which trophic structures are developed. Thus, an alteration in the litter or resource base is likely to affect the decomposer community and consequent trophic levels (bottom-up effects). The use of Oribatida as biological indicators can also be applied to evaluate any impact on the mite communities.

The Native Jarrah Forest Ecosystem

Firstly, the mite communities under native *E. marginata* (jarrah) forest in southwestern Australia will be used as the benchmark for evaluating the disturbed ecosystems. Given the paucity of information about the soil and litter mite communities of the jarrah forest ecosystem, this study adds significantly to other studies that have examined the soil and litter fauna of the jarrah forest. Since very few of these studies consider the lower taxonomic levels, the classification of mites to the family, genus and species levels in the present study adds significance to the ecological investigation of the jarrah forest. Previous

studies of the soil and litter fauna of jarrah forest include data on general invertebrate abundance (Springett, 1976a & b; Postle, 1989), species composition (Springett, 1976b; Postle, 1989) and seasonality (Koch and Majer, 1980; Majer and Koch, 1982; Postle, 1985). Others have examined the influence of major impacts to the jarrah forest soil and litter fauna, such as prescribed burning (McNamara, 1955; Springett, 1979; Abbott, 1984; Majer, 1984), bauxite mining (Majer, 1980) and the effect of the fungus *Phytophthora cinnamomi* (Postle, Majer and Dell, 1986). The rehabilitation of bauxite mining sites has also been investigated (Cuccovia, 1997).

At the class and sub-class levels we have good knowledge of the diverse taxonomic groups of invertebrates that inhabit the jarrah forest (Abbott, 1984; Major, 1984; Postle, 1985; Postle *et al.*, 1986; Major and Abbott, 1989). The soil and litter fauna show considerable diversity and abundance considering the jarrah forest has undergone dramatic modification due to unsustainable logging of old growth trees, modification of fire regimes, mining activities and the introduction of *Phytophthora cinnamomi*. In view of the evolutionary history of the region and the fact that great assemblages of phytophagous insect species have coevolved with and are dependent upon *Eucalyptus* (Florence, 1996), invertebrate communities appear to tolerate degrees of disturbance to the forest. For example, invertebrate communities have a remarkable capacity to survive an ecological disturbance such as fire and/or are able to recolonise burnt sites rapidly. Both Majer (1984) and Abbott (1984) reveal negative short-term effects of fire in jarrah forest on soil and litter animals, followed by quick recovery for most taxa. Campbell and Tanton (1981) found recovery occurred within 18 months in eucalypt forest in southeastern Australia, and Neumann (1991) found a broad mix of predators, parasitoids, carrion and seed-feeders, foragers and

omnivores active immediately after wildfire in *E. regnans* regrowth forest (including mites).

The mites and springtails numerically dominate the soil and litter layers of the jarrah forest as they do in forests elsewhere (Ahern and Yen, 1977; Plowman, 1979; di Castri and Vitali-di Castri, 1981). Of the total soil mesofauna inhabiting the jarrah forest of the present study the mites comprised between 86 – 99% and the springtails 1 – 14%. In the litter layer, mites comprised 65 – 77% and the springtails 6 – 20 % of the total mesofauna. Other authors (Ahern and Yen, 1977; Majer, 1984; Springett, 1976) have reported similar abundances. In the jarrah forest, Majer and Abbott (1989) found mite abundances made up 92.6 % of the total soil invertebrates and 77.5 % in the litter layers, and Postle (1989) found mites and springtails comprised over 90 % of the invertebrate fauna in both the soil and litter layers.

Total mesofaunal abundances in the present study ranged between 290 - 2800 kg⁻¹ of dry litter and $74 \times 10^3 - 114 \times 10^3 \text{ m}^{-2}$ of soil, with mite abundances of between 194 - 2193 kg⁻¹ in the litter and soil abundances similar to the total soil mesofauna. Generally, the soil values were considerably higher than those recorded for soil invertebrates in other jarrah forest studies and in other Australian mediterranean-type forests. For example, Postle (1985) recorded soil invertebrate density between $39 \times 10^3 - 134 \times 10^3 \text{ m}^{-2}$ and in his 1989 study between $43 \times 10^3 - 61 \times 10^3 \text{ m}^{-2}$. Majer (1984) and Cuccovia (1997) recorded similar soil mite abundances, approximately $3 \times 10^3 - 25 \times 10^3 \text{ m}^{-2}$ and $25 \times 10^3 \text{ m}^{-2}$, respectively. Hutson and Veitch (1983) recorded between $25.7 \times 10^3 - 34.9 \times 10^3 \text{ m}^{-2}$ of mites in the upper soil layers of three South Australian low open forests. Variations in abundances between these studies may be attributable to differences in sampling techniques and timing

of sampling. Litter abundances are difficult to compare due to the differences in data formats. For example, abundances have been expressed per unit volume, per unit area or per unit of dry weight of litter.

The large seasonal variation in mesofaunal abundances and diversity is not unusual (Majer and Abbott, 1989; Postle, 1989) and is most likely associated with soil moisture. The higher abundance and diversity of soil and litter mesofauna recorded in autumn were associated with higher soil moisture values than those recorded in the spring sampling period.

According to di Castri and Vitali - di Castri (1981) seasonal changes in water availability are a major factor in inducing population changes. The commencement of the wet season, thus increasing soil moisture, initiates litter decomposition, which in turn stimulates a subsequent increase in population densities (Crossley and Hoglund, 1962; di Castri and Astudillo, 1966, as cited in di Castri and Vitali - di Castri, 1981; Plowman, 1979; Postle, 1989). The significantly higher abundances in autumn and increase in diversity in this study is due mainly to an increase in the number of larval mites, also noted by Postle (1989) in his study. Exceptionally high abundances of mesostigmatid nymphs and high abundances of prostigmatid (Eupodoidea) nymphs were noted in all sites sampled in autumn. Further, the jarrah forest in autumn displays a diverse mycoflora. Throughout this season there is a succession of fungal genera and species that develop in the various substrates of the forest, reflecting substrate quality and associated microclimatic conditions (Hilton, Malajczuk and Pearce, 1989), thus attracting a diverse mesofauna.

The relative contribution of the four acarine orders within the jarrah forest of the present study is, at first glance, unusual for a temperate forest ecosystem. In other such forests in

Australia the Oribatida are numerically dominant followed by the Prostigmata, Mesostigmata and finally the Astigmata (Spain and Hutson, 1983; Springett, 1976a). In the few local studies, in *Eucalyptus*, *Banksia* and *Hakea* woodland at Gnangara (Springett, 1976b) and in the jarrah forest (Postle, 1989; Cuccovia, 1997), oribatids were the most abundant group of mites. They were also the dominant order in the litter at a site with similar mediterranean climate in South Australia (Hutson and Veitch, 1983) and within the sclerophyll forest in di Castri and Vitali – di Castris' (1981) Chilean study. In this study, although their abundance was far surpassed by the Astigmata, oribatids were the next dominant order in spring.

The substantially high numbers of Astigmata in the jarrah forest is uncommon since these mites are typically found in very low numbers in forests, grasslands and other undisturbed ecosystems (Phillips, 1990). This particular order was represented by a single morphospecies, *Tyrophagus*, a cosmopolitan genus that invades soil and litter habitats (Krantz, 1978; O'Conner, 1982). Cleared lands surround the jarrah forest of this study, and their close proximity may allow *Tyrophagus* to effectively invade and colonize the forest ecosystem. Such large numbers of *Tyrophagus* are usually a result of their high fecundity and effective dispersal stage (Walter, Hudgens and Freckman, 1986).

The remarkable species richness and the large numbers of rare locally endemic plants in southwest Australia are a result of long and complex evolutionary processes, stimulated by the environmental perturbations of increasing aridity and fire coupled with resource (nutrient and water) scarcity (Hobbs *et al.*, 1995a). The floristic density and diversity of the jarrah forest understorey provide diverse habitats within the litter layer, supporting mite

populations of high abundance and diversity. The various woody, reproductive and other litter fractions all have different decomposition dynamics (Spain and Hutson, 1983), contributing to the diversity of the organic matter in the forest litter. The open canopy of the jarrah forest and its associated uneven spread of the litter layer allows a large percentage of sunlight and rainfall to reach the surface litter and soil surface, contributing to high soil and litter temperatures and higher soil moisture values than would be recorded under forest of closed canopy.

Given this, how do the mite communities of this area compare in species richness and species composition with other forest areas of Australia and the world? There have been few comparable intensive studies of the mite fauna of other Australian or mediterranean forest ecosystems. In this study 48 families and 95 morphospecies/species were identified from the soil and litter layers of the jarrah forest, the highest recorded thus far for this ecosystem. Koch and Majer (1980) collected only 3 families in their investigation of the jarrah forest, with no species identified, and Cuccovia (1997) identified 28 families and 49 morphospecies in his investigation of mite succession in revegetated forest sites. In the dry sclerophyllous *Eucalyptus* and *Hakea* woodland at Gnangara Springett (1976) recognized 9 families and 43 morphospecies/species. It is difficult to highlight the numerically important families of the jarrah forest, due to the limitation in taxonomic identification in these other studies. Notwithstanding this, several families are notable for their presence in each study, particularly the Oribatida and Prostigmata. Oribatid families well represented include Oppiidae, Oribatulidae, Haplozetidae and Brachychthoniidae, with Galumnidae, Mycobatidae and Ceratozetidae appearing in lower abundances and species richness. Well-represented Prostigmata include Eupodidae, Pseudocheylidae, Rhagidiidae, Stigmaeidae,

Bdellidae and Cunaxidae. The Mesostigmata and Astigmata have not been identified beyond the ordinal level in these other studies thus no comparison can be made.

The mite communities of the native jarrah forest provide a baseline against which to evaluate the impacts of plantation development on mite assemblages in southwestern Australia. Given the chronological sequence of events this study demonstrates, it is necessary to interpolate the pasture ecosystem, both as the other extreme on the disturbance gradient and as the 'starting point' of the *E. globulus* revegetation.

The Pasture Ecosystem

The native jarrah forest and pasture ecosystems can be viewed as two extremes of a disturbance gradient. The conversion from native forest to pasture involves both niche reduction and alteration of the ecosystem processes (Tian, Brussaard, Kang and Swift, 1997). The absence of trees and understorey plant species, and the minimal groundcover in the pasture environment substantially reduces the amount and diversity of organic matter reaching the soil. Since litter operates as a substrate that facilitates other organisms (Grime, 1979; Facelli and Pickett, 1991; Facelli and Kerrigan, 1996) and is an essential reservoir of nutrients for plant and microbial growth (Taylor *et al.*, 1989; Tamm, 1991; Northup *et al.*, 1995, as cited in Nilsson, Wardle and Dahlberg, 1999), its absence significantly reduces the resource niche. Consequently, the decomposer community is dramatically affected and the saprophagous mite population reduced.

The buffer provided by trees and understorey vegetation against soil microclimate fluctuations is significantly lost in the pasture, resulting in a harsh environment with large extremes in soil moisture and temperature. These two variables (moisture and temperature) have fundamental effects on decomposition, significantly controlling decomposer activity (Swift *et al.*, 1979). Increased sunlight reaching the soil surface increases soil temperatures (Crawley, 1983), with resultant low soil moisture content due to increased evaporation at the soil surface. The combination of reduced plant rooting depths and nodes, and a higher sand component would increase soil erosion and facilitate rapid draining of water through the soil, resulting in a reduction in water retention capacity and increased leaching of nutrients (Hobbs *et al.*, 1995a).

Despite the increasing number of studies assessing the impact of agriculture on invertebrate communities, very few pasture sites have been studied worldwide. There have, however, been several studies on the impact of livestock grazing. Livestock grazing significantly alters the composition of soil mesofauna. King and Hutchinson (1976) suggest that the large effects of sheep grazing intensity on soil mesofaunal abundance are due mainly to changes in habitat, such as increased removal of herbage by sheep. Consequently, the amount, type and distribution of food for the mesofauna decrease as stocking intensity increases (Hutchinson and King, 1980). King and Hutchinson (1983) also noted that soil compaction as a result of livestock grazing can increase soil pH through poor aeration (in contrast to a lower pH of the pasture in the present study), and may contribute to reduced mesofaunal numbers when compared with native grasslands. Soil mite abundances in the pasture ranged between $41 \times 10^3 - 492 \times 10^3 \text{ m}^{-2}$, which are substantially higher than those recorded by King and Hutchinson (1983) who found the accumulative effects over time of

sheep grazing at 5 sheep ha⁻¹ decreased soil mite abundances to 4500 m⁻². In their 1976 study, King and Hutchinson noted abundances ranging between 17.4 x 10³ m⁻² and 23.7 x 10³ m⁻² under pasture grazed at 10 sheep ha⁻¹.

The majority of mites extracted from the pasture were Mesostigmata and Astigmata, members of which are well able to tolerate disturbance (Hudson, 1980; Tilman, 1996). Astigmata populations increase significantly in disturbed and human-modified ecosystems, including grazed pastures. They are considered to be effective opportunistic colonizers, being specialists in exploiting spatially and/or temporally restricted microhabitats aided by their small size, short generation time and presence of an effective dispersal stage (Hudson, 1980; Philips, 1990; Behan-Pelletier, 1999). The addition of fertilizer and/or livestock urine and faeces in the grazed pasture stimulates an increase in microorganism populations, which in turn stimulates an increase in Astigmata (Philips, 1990; Lussenhop, 1992). These microorganisms are considered potential food sources. The Acaridae (including *Tyrophagus*) in particular, are positively influenced by agricultural disturbance (Crossley *et al.*, 1992). Since the Mesostigmata are known to feed on smaller astigmatic mites (Crossley *et al.*, 1992), an increase in the abundance of Astigmata could stimulate an increase in the abundance of predatory Mesostigmata.

The Prostigmata, in particular Eupodidae and Tarsonemidae, are known inhabitants of disturbed soils, being typically more numerous in agricultural soils (Lagerlof and Andren, 1988; Crossley *et al.*, 1992; Tian *et al.*, 1997). These families respond rapidly to disturbance such as plowing and cultivation, and fertilizer use may encourage multiplication (Behan-Pelletier, 1999).

No oribatid mites were extracted from the pasture in the present study, although this does not necessarily indicate that they are not present. They may not have been extracted because of very low abundances. Oribatid populations decline rapidly when their habitat is damaged due to their generally low metabolic rates, slow development and low fecundity (Behan-Pelletier, 1999). Hence, they are characteristic of undisturbed soils with good litter cover, high organic residues and high relative humidity. Oribatid populations in the pasture are inhibited simply because the environmental conditions are not suitable. The limited organic matter provides a depleted resource base, and impoverished microhabitat and microclimate. Temperature and moisture extremes have a direct effect on the physiological activity of oribatids as reflected in respiration, ingestion, growth and survival (Mitchell, 1979).

Reduced species diversity can be due to either or both a reduction in the number of species and/or a distortion in the equitability component of species diversity (Lloyd and Ghelardi, 1964, as cited in di Castri and Vitali-di Castri, 1981), where a few species become extremely abundant relative to the others. This redundancy, as it is termed, was especially high in the grazed pasture due to the strong dominance of only a few species (*Phytoseiidae* and *Tyrophagus*). Di Castri and Vitali-di Castri (1981) also noted this contraposition between decreasing species diversity and increasing density in fields under irrigation. The few mite species identified from the grazed pasture were found to be within the same functional groups, a topic I shall return to later.

Conversion of native forest to a long-term pasture shows a reduction in resource niche, microhabitat and microclimate. The resultant impoverished ecosystem has a reduced decomposer community, which in turn influences the higher trophic levels.

The *Eucalyptus globulus* Plantation Ecosystem

The *E. globulus* plantation can be viewed from several perspectives. Firstly, the re-vegetation of a long-term grazed pasture with a monoculture *E. globulus* results in a more complex habitat for the mite community. The almost closed canopy of the *E. globulus* plantation and the evenly distributed litter layer provides a buffer against fluctuations in the soil-litter microclimate. Litter and soil temperatures and soil moisture values remain low mainly due to the canopy and litter intercepting sunlight and rainfall from reaching the soil surface. The substantial litter build-up, exhibiting considerable depth at the base of trees, and the presence of fallen trees and branches within the plantation has also increased both the microhabitat niche and resource base. As a re-vegetation succession of 8 years of age, what has been the impact of this increasing habitat complexity on the soil-litter mesofauna? However at the same time, the single species of litter produced from the *E. globulus* plantation provides a less diverse, complex environment than the native jarrah forest, and is still therefore an order of magnitude of disturbance away from the original forest ecosystem. Combined with the absence of an understorey in the plantation, the monospecific litter would produce a limited range of nutritional components (Chapman, Whittaker and Heal, 1988) and a less diverse organic matter, thus providing a narrowed resource niche. Therefore, to what extent have the mite assemblages moved toward the complexity of those of the native forest?

How do the mite communities of the monoculture plantation compare with the pasture and native jarrah forest? As expected, the mites and springtails numerically dominated the litter layer and soil under the *E. globulus* plantation, ranging in abundance between 180 – 2900 kg⁻¹ of dry litter and 75 x 10³ – 196 x 10³ m⁻² of soil, with mite abundances in the litter ranging between 34 – 1867 kg⁻¹, and soil abundances similar to the soil mesofauna. There is limited information from other studies on the relative number of mesofauna recorded in the soil and litter of *E. globulus* monocultures, and any comparisons are difficult due to the differences in the expression of abundances and the methodology used. Generally, mesofaunal abundances were similar in the surface litter of the native forest and *E. globulus* plantation. However, when Serralheiro and Madeira (1990) compared densities of arthropod populations under an *E. globulus* plantation with those under native vegetation dominated by *Quercus suber* in central Portugal, density was significantly lower in the forest floor (litter) of the plantation (2303 – 19793 m⁻²) than the native vegetation (7235 – 37201 m⁻²). Soil mesofaunal abundances under the present *E. globulus* plantation were higher than under the native forest, yet were lower than the pasture in the autumn samples. This trend is comparable to the mean soil arthropod densities (individuals per dm³) extracted from human-modified and natural ecosystems in Chile (di Castri and Vitali-di Castri, 1981), where mesofaunal densities were slightly higher in the *E. globulus* plantation than in the sclerophyll forest. In contrast, Serralheiro and Madeira (1990) noted slightly greater soil abundances under the native *Q. suber* forest (585 – 5822 dm³) than under the *E. globulus* plantation (344 - 4453 dm³).

Mite abundances in the surface litter were greater under the jarrah forest than under the monoculture *E. globulus* plantation, yet soil abundances were greater under the *E. globulus* plantation. Di Castri and Vitali – di Castri (1981), however, noted mite densities to be very similar in soil under the sclerophyll forest and the *E. globulus* plantation, at 1067.6 and 1070.0 ind. dm³ respectively.

Considering the previous land use (as pasture) of the *E. globulus* plantation and the disturbance both the pasture and plantation ecosystems have experienced, the great abundance of disturbance-tolerant *Tyrophagus*, as well as mesostigmatid and prostigmatid mites was perhaps to be expected in the plantation sites. In contrast, di Castri and Vitali – di Castri (1981) found the Astigmata, Mesostigmata and Prostigmata all with higher abundances under the sclerophyllous forest compared to the *E. globulus* plantation in their Chilean study, though the history of the plantation is not disturbed. Oribatid abundances in the *E. globulus* litter were lower than in the jarrah forest litter, yet soil abundances were greater under the plantation. This result is similar to that found by Serralheiro and Madeira (1990) who noted that the forest floor (litter) density was significantly greater under the native vegetation than under the *E. globulus* plantation. Di Castri and Vitali – di Castri (1981) also noted higher oribatid densities in the soil under the *E. globulus* plantation than in the sclerophyllous forest.

It is well understood that reduced biological diversity is a result of simplification of ecosystem structure and function (Hobbs *et al.*, 1995a) and is especially known to occur in manipulated or disturbed ecosystems (di Castri, 1973). Mite species richness of the *E. globulus* litter layer and soil was substantially lower than that of the jarrah forest, probably

a reflection of the monospecific litter environment or longer colonisation time or other factors. Species diversity in the litter was notably greater in the jarrah forest, reflecting the more diverse microhabitat and resource niches. The lower diversity recorded in the *E. globulus* litter was also noted by di Castri and Vitali-di Castri (1981) in their Chilean study. Surprisingly, species diversity in the soil was slightly higher under the *E. globulus* plantation than under the jarrah forest, and it may be that differences caused by the different litter types may not be reflected at the soil level or there is a more continuous distribution of true soil fauna.

The resultant trend of mite species richness follows the general trend of invertebrate species richness in Mediterranean ecosystems (Postle, 1985; Legakis, 1994). The abundance of mites is higher in the soil than in the surface litter, and diversity is higher in the litter. Also, less disturbed sites are richer in species than more disturbed sites and more diverse habitats are richer in species than more uniform ones (di Castri, 1973; Gross, 1985 as cited in Legakis, 1994). The less disturbed *E. globulus* plantation was more species rich than the more disturbed pasture, and the highly diverse native forest had higher species richness than the monoculture plantation.

Although the *E. globulus* plantation was planted in 1990, re-establishment of mite communities to a level of diversity similar to the adjacent native jarrah forest may take long periods of time, if it occurs at all! Land use changes, especially of grazed pasture to plantations, affect the quality of soil organic matter and thus the amount and rate of net N mineralization (Aggangan, O'Connell, McGrath and Dell, 1998). Changes associated with the development of a litter layer are complex due to microclimate factors, as well as the

form, availability and chemical release of litter nutrients (Hilton *et al.*, 1989). These changes in the physical and chemical components ultimately influence the ecology of the ecosystem.

Initially, it would be expected that the more disturbance-tolerant mite species, such as members of Astigmata, Prostigmata and Mesostigmata, would dominate the soil-litter system of the newly revegetated land (Tilman, 1996; Beckmann, 1988 and Skubala, 1995, as cited in Behan-Pelletier, 1999).

The oribatids increased in species diversity with increasing vegetation complexity, along the habitat gradient monoculture *E. globulus* plantation to the heterogeneous jarrah forest. The heterogeneous nature of the litter of the jarrah forest provides a wider and more diverse niche for the oribatids. It has been demonstrated that more complex habitats accommodate more diverse mite faunas than simple habitats. Small-scale habitat changes on the forest floor, from litter to lichen and moss to mushrooms, have been related to changes in oribatid species composition and hence contribute to oribatid diversity (Hunt, 1994). Cuccovia (1997) noted that as the ages of rehabilitated vegetation sites increased, producing an effective canopy and litter layer (increasing soil organic carbon), the sites became more favourable for the oribatids. Cuccovia also found species richness of the oldest rehabilitation site of 20 years remained well below the richness of the ancient native forest. Hansen and Coleman (1998) and Hansen (2000) also demonstrated a link between litter habitat heterogeneity and diversity of oribatids; species diversity declined in habitats of reduced heterogeneity (monotypic litter).

It is expected that plant diversity can affect belowground diversity, given that soil faunal and microbial diversity can be related to microhabitat heterogeneity. Given that oribatids tend to favour stable and regulated habitats, their assemblages under the *E. globulus* plantation (although lower in species diversity) suggests establishment of some degree of suitable microclimate, microhabitat and resource niche at least 8 years after planting. As the plantation system matures, numbers of fungal species available for fungivores may increase. For example, Lu, Malajczuk, Brudrett and Dell (1999) found an increase in the number of fungal species (12-17 per site) in the 6 – 8 year old *E. globulus* plantation of their study, yet richness was lower than that observed in the native eucalypt forests. The increase in litter build up as canopy closure becomes complete may facilitate an increase in the number of fungal species and the slow recolonization of oribatid mite populations. The close proximity of the plantation to the jarrah forest would allow for fungal recolonization primarily by means of spore dispersal via wind from the adjacent forest (Lu *et al.*, 1999).

The most represented families of oribatids under the *E. globulus* plantation are common to disturbed habitats. Species in the families Tectocepheidae (i.e. *Tectocepheus velatus*) and Oppiidae generally are common to agricultural soils and heavily disturbed or early successional habitats, exhibiting high levels of ecological tolerances or ‘insensitivity’ (Dindal, 1977; Aoki, 1979; Rockett, 1986; Franchini and Rockett, 1996; Cuccovia, 1997; Behan-Pelletier, 1999).

Functional Group Structure

The simplified ecosystems of the monoculture *E. globulus* plantation and grazed pasture are reflected in the structure of the soil-litter mite community, and can be illustrated using the concept of functional groupings. The trophic roles of microarthropods in belowground detrital food webs are increasingly represented by functional groups (Mueller *et al.*, 1990; Brussaard *et al.*, 1997). This concept offers a first approximation of species that may be ecologically linked (Walter and Proctor, 1999), lacks the assumption about the importance of competition among members, and does not require knowledge of taxonomic affinities (Moore, Walter and Hunt, 1988; Crossley *et al.*, 1992). Functional groups take into account how the use of resources by one group may change their availability for others (Walter and Proctor, 1999).

Each functional group is defined by the resources used and is based upon the trophic level and the feeding method (Moore *et al.*, 1988; Walter and Proctor, 1999). Microarthropods can be classified into the functional groups of predators, phytophages [specifically macrophytophages (feeding strictly on higher plant material), microphytophages (feeding on specific microfloral elements: fungi, bacteria, yeasts and algae), panphytophages (those that utilize both microbial and higher plant materials)], saprophages (those that feed on detritus and living microbes) and parasites (Moore *et al.*, 1988; Norton, 1990; Walter and Proctor, 1999).

The use of functional groups in the present study is a useful exercise, as it allows further insight into the impact of re-vegetation on the soil-litter mite communities. Given that the

resource niche is expanded with re-vegetation, the influence this has on the trophic structure can be significant. The assignment of mites to functional groups (microphytophagous, phytophagous, predator) (Table 4.1) in this study is based upon the literature; the time required to observe the feeding habits of each mite species would be phenomenal! However, I am aware of the inherent problems when assigning mite families to a functional group without specific knowledge of feeding preference. Also, membership in a particular functional group is not necessarily exclusive – an organism may belong to more than one group over its lifetime (Walter and Proctor, 1999). For these reasons, some caution is required when using information presented in Table 4.1.

In soils with depleted organic resources, such as many human-modified ecosystems, food webs tend to have reduced numbers of trophic levels and fewer components at each level (Swift and Anderson, 1993; Lavelle, 1996). As might be expected from the greater species diversity, the native jarrah forest had the greatest number of orders in the microphytophagous and phytophagous functional groups compared to the *E. globulus* plantation (Figures 4.1 and 4.2). If the plantation ecosystem has fewer different resources, and thus fewer niches, it would support fewer species than the jarrah forest with its greater variety of resources (Pianka, 1975). The reduced complexity of the monoculture plantation ecosystem supported fewer families (and species) in each functional group. Since species vary in the amount and type of contribution they make to an ecosystem process, loss of a species may be compensated for by another functionally similar species (Freckman, Blackburn, Brussaard, Hutchings, Palmer and Snelgrove, 1997). Although none of the

Table 4.1 Possible functional groupings of mite families found in the soil-litter subsystem of a native *E. marginata* forest, an 8 year-old *E. globulus* plantation and a grazed pasture.

Functional group	Family	References
Microphytophagous	Nanorchestidae	Walter, 1988; Kay <i>et al.</i> , 1999
	Eupodidae	Krantz, 1978; Crossley <i>et al.</i> , 1992; Walter and Proctor, 1999
	Uropodidae	Krantz and Ainscough, 1990
	Astegistidae	
	Brachychthoniidae	
	Ceratozetidae	
	Cymbaeremaeoidea	
	Damaeidae	Norton, 1990
	Eremaeozetidae	
	Haplochthoniidae	
	Haplozetidae	Mueller <i>et al.</i> , 1990
	Humerobatidae	
	Mycobatidae	
	Oppiidae	Norton, 1990
	Oppiinae	Norton, 1990
	Oppiellinae	Norton, 1990
	Phenopelopidae	
	Plateremaeidae	
	Scutoverticidae	
	Scheloribatidae	
	Spherochthoniidae	
	Tectocepheidae	Schenker, ?; Skubala, 1997
	Tarsonemidae	Krantz, 1978; Kethley, 1990; Mueller <i>et al.</i> , 1990; Walter and Proctor, 1999
Phytophagous	Genus <i>Tyrophagus</i>	Philips, 1990
	Laelapidae	Knihinicki and Halliday, 1995
	Phytoseiidae	Krantz and Ainscough, 1990; Walter and Proctor, 1999
	Cryptognathidae	Luxton, 1973; Krantz, 1978
	Linotetranychidae	
	Penthalodidae	Walter and Proctor, 1999
	Stigmaeidae	Krantz, 1978; Walter and Proctor, 1999
	Tarsonemidae	Krantz, 1978
	Tetranychidae	Baker and Pritchard, 1953; Krantz, 1978; Kethley, 1990
	Galumnidae	Norton, 1990
	Oribatulidae	Norton, 1990

Table 4.1 *continued*

Functional group	Family	References
Predator	Ascidae	Koehler, 1999
	Eviphididae	Koehler, 1999
	Phytoseiidae	Koehler, 1999; Walter and Proctor, 1999
	Ologomasidae	Koehler, 1999
	Laelapidae	Cassanueva, 1993; Walter and Proctor, 1999
	Uropodidae	Walter and Proctor, 1999
	Genus <i>Tyrophagus</i>	Philips, 1990
	Anystidae	Krantz, 1978; Kethley, 1990; Walter and Proctor, 1999
	Bdellidae	Krantz, 1978; Kethley, 1990; Walter and Proctor, 1999
	Caeculidae	Krantz, 1978; Kethley, 1990; Walter and Proctor, 1999
	Camerobiidae	Gerison, 1972; Krantz, 1978
	Cheyletidae	Krantz, 1978; Gerison, 1994
	Cryptognathidae	Luxton, 1973
	Cunaxidae	Smiley, 1975; Krantz, 1978; Kethley, 1990; Walter and Proctor, 1999
	Erythraeidae (adult)	Krantz, 1978; Fain, 1991; Lo, 1996; Walter and Proctor, 1999
	Eupodidae	Crossley et al., 1992
	Pachygnathidae (=Bimichaelidae)	Walter, 1988
	Pseudocheylidae	VanDis and Ueckermann, 1991; Walter and Proctor, 1999
	Raphignathidae	Krantz, 1978
	Rhagidiidae	Krantz, 1978; Walter and Proctor, 1999
	Smarididae (adult)	Krantz, 1978; Zhang, 1995

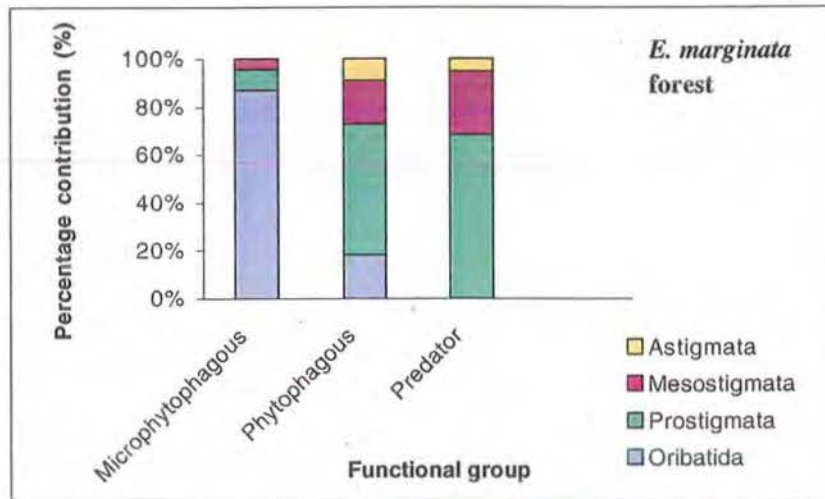


Figure 4.1. Percentage contributions of acarine orders within each functional group in the soil-litter component under a native *E. marginata* forest.

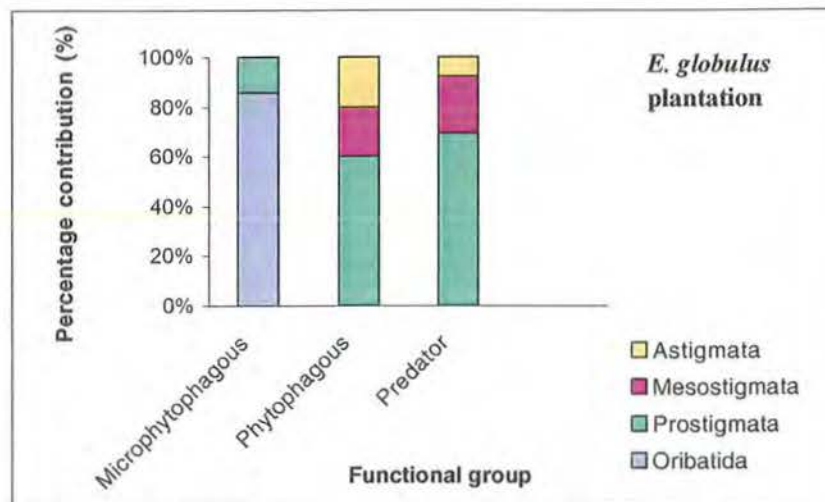


Figure 4.2. Percentage contributions of acarine orders within each functional group in the soil-litter component under three 8 year-old *E. globulus* plantations.

distinguished functional groups were absent from the plantation, no oribatid families were present at the phytophagous level.

In comparison to the native forest, the *E. globulus* plantation had a lower number of oribatid families at the microphytophagous level. There were, however, large populations of Oppiidae, Oppiinae, Oppiellinae and Tectocepheidae within these plantation sites. Fungi comprise the bulk of the diet of Oppiids and their significant presence suggests an opportune resource niche under the plantation.

The Phytoseiidae and genus *Tyrophagus* were both present under the *E. globulus* plantation and were both placed in the phytophagous and predaceous functional groups. The Phytoseiidae are known to supplement their arthropod diets with fungi, as well as pollen and plant nectar (Krantz, 1978; Walter and Proctor, 1999). *Tyrophagus* show varying degrees of selectivity in choosing fungi for consumption (Krantz, 1978) and are also known to prey on nematodes (Walter, Hudgens and Freckman, 1986).

The number of recognized predaceous families was slightly lower under the *E. globulus* plantation compared to the native forest. Typical predaceous families were represented, such as the prostigmatids Stigmaeidae, Bdellidae, Cunaxidae, Cheyletidae, Erythraeidae, Eupodidae and Rhagidiidae, and the mesostigmatids Phytoseiidae, Ologomasidae and Ascidae. Prey for these mites include nematodes, arthropod eggs, phytophagous mites and insects (Walter and Proctor, 1999).

The grazed pasture supported the smallest number of functional groups (Figure 4.3), with the smallest number of families in each group. The disappearance of functional groups is common in agricultural soils (Swift and Anderson, 1993), and was evident here with the absence of the microphytophagous mites, particularly Oribatida, from the pasture ecosystem. Reasons for this have been discussed. Phytoseiidae and *Tyrophagus*, also well represented under the pasture, were both placed in the phytophagous and predaceous functional groups. Although fungal biomass associated with eucalypts under the pasture is likely to be limited, the annual application of fertilizers may encourage microfloral populations to thrive, stimulating *Tyrophagus* and nematode populations. The number of predaceous families identified under the grazed pasture was lower than under the *E. globulus* plantation, yet still included members of the major predator families Phytoseiidae, Bdellidae, Cunaxidae, Erythraeidae and Rhagidiidae. Besides the physical disturbance in the pasture, the reduced number of predators can be related to the lack of a litter layer, which does not allow the development of a diverse prey spectrum to feed these predators (Koehler, 1999).

Simplifying the vegetation of an ecosystem not only changes the physical habitat- it is also likely to reduce the breadth of the resource base available to the primary decomposers. For example, a reduction or alteration in the diversity of the microfloral populations may have bottom-up effects through to the saprophagous mites, reducing both species and functional group diversities (Wardle and Giller, 1997; Scheu and Schaefer, 1998). Oribatid mites are the primary mite constituents of the saprophagic functional group (Walter and Proctor, 1999). The lower number of oribatid families under the plantation may be a consequence

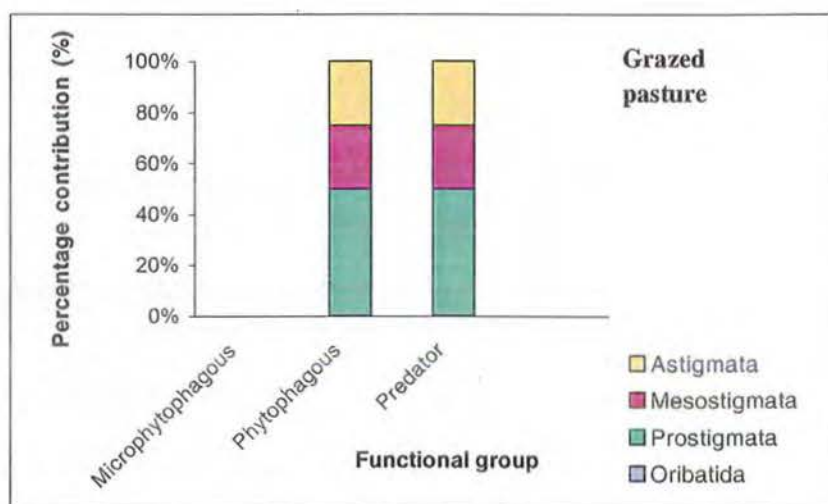


Figure 4.3. Percentage contributions of acarine orders within each functional group in soil under a grazed pasture.

of the narrowed resource niche, particularly fungal diversity. Oribatid families that were present in the native jarrah forest, yet were absent from under the plantation include Astegistidae, Brachychthoniidae, Ceratozetidae, Cymbaeremaeoidea, Damaeidae, Galumnidae, Eremaozetidae, Haplozetidae, Humerobatidae, Oribatulidae, Phenopelopidae, Plateremaeidae, Scheloribatidae, Scutoverticidae and Spherochthoniidae. The reduced number of families in the phytophagous and predator functional groups under the plantation is another possible bottom-up effect of resource limitation. The reduced number of microfloral and detrital feeding mites may reduce the number of predators preying on these mites.

It is reasonable to expect that the monospecific litter attracts a less diverse microflora, and the presence of indigenous fungi under the *E. globulus* plantation is most likely limited given the history of the site. The more uniform environment may also be limited in the diversity of tree root-fungal associations compared to the jarrah forest where many major tree and shrub species are ectomycorrhizal with associations probably involving hundreds of interacting fungal species. A reduction in the diversity of indigenous fungi and thus a reduction in fungal resources may in turn affect mite diversity. For example, oribatid numbers and richness have been correlated with decomposition rate within simple litters (Hansen and Coleman, 1998). More so, introduced fungal species may colonize and increase in biomass, further affecting feeding as there is ample evidence to suggest that mesofauna selectively graze (McLean, Kaneko and Parkinson, 1996). McLean *et al.* (1996) also note that mesofauna are known to shift from a preferred to a less preferred fungal species as the preferred species is grazed down. Adapting to a change in introduced fungal species could selectively permit particular species to survive and reproduce in that

particular habitat. It is not known whether the *E. globulus* seedlings were inoculated with spores of ectomycorrhizal fungi before planting, as this may further influence any interactions with indigenous fungi and subsequent soil fauna.

Shortcomings of the Study

As previously discussed the pseudoreplicate design of the study reduced the number of statistical analyses used. Seasonal samples were taken at different sites to reduce the impact of soil cores, since they are considered a disturbance to the soil-litter stratum. As a result, replication was reduced, limiting true seasonal comparisons within vegetation types and sites. Also labour-intensive and time-consuming nature of species-level identification, together with the limitation in mite taxonomy, was a significant limitation of this study, particularly with its own time constraints.

Confounding variables of the *E. globulus* plantation and pasture ecosystems exist. Since the pasture is an environment that was man-created after being a long-lived natural ecosystem and the plantation was planted on ex-pasture, it is difficult to exclusively separate the effects of the plantation from the pasture. Very few studies have investigated the effects of a previous land use on a current ecosystem, particularly *E. globulus* plantations. Aggangan, O'Connell, McGrath and Dell (1998) studied the influence of ex-pasture and ex-native forest sites on the amount and rate of net N mineralization in *E. globulus* plantations in Western Australia. Results indicated that previous land use and fertilization with N and P have a significant influence on both the amount and rate of net N mineralization and

microbial respiration in these current *E. globulus* plantations, modifying the soil P status and the quantity and quality of organic matter.

The effect of fertilization in the *E. globulus* plantation is difficult to separate from the effect of fertilization on the previous land use. Both the *E. globulus* plantation and grazed pasture topsoil were slightly more acid than the soil under the jarrah forest, possibly a result of the long history of repeated application of fertilizer treatments containing phosphate and nitrates into the pasture prior to plantation establishment. The more alkaline soils under the native jarrah forest may be attributed to the prevention of substantial humus accumulation by recurrent bushfires, such as the 1993 fuel reduction burn, and this in combination with recycling of calcium and potash by trees (Havel, 1975).

Issues Arising from the Study

It is not possible to say that the planting a monoculture *E. globulus* plantation has affected the functioning of ecosystem processes, such as decomposition and nutrient cycling. More in depth studies are required to determine this. Understanding the effects of changes in food web structure or changes in the pattern of the transfer of matter within the food web, and how these changes affect decomposition and nutrient cycling (Moore and de Ruiter, 1991; Baskin, 1994) is one issue. For example, the loss of any particular species may seem to have no direct impact on soil quality, yet it may severely affect those species with more direct roles through food web interactions (Stork and Eggleton, 1992). The identification of functional groups that occupy key positions in ecosystem processes *in situ* is also important. The loss of a functional group, composed of only a few species, could have detrimental

effects on the ecosystem, particularly in environments where diversity is limited (Freckman *et al.*, 1997). What is the minimum number of functional groups, and species within functional groups, to ensure soil resilience against ecosystem stress and disturbance?

Further studies could also ask, as the plantations mature, does mite diversity vary over time and space? What is the length of time of recolonization of indigenous and cosmopolitan mite species into the monoculture plantation?

Management Implications of *Eucalyptus globulus* Plantations

The extensive establishment of these plantations in southwest Western Australia within an existing agricultural landscape and at the expense of native forest has the potential to reduce the biodiversity of the region. The establishment of *E. globulus* plantations could benefit biodiversity if planted on pasture, yet would create the opposite effect if native forest were converted to plantation. To minimize the impact of these plantations on endemic native fauna and flora, conservation of the existing native forests is extremely important, as these areas function as refugia or corridors of diversity. Objectives need to include the maintenance of a minimum diversity of plants that provide the diversity and abundance of resources necessary to sustain a complex community (Lavelle, 1996). This may include the maintenance of key functional groups.

Moreover, if it were unavoidable that we compromise areas of the native forests of Western Australia, it would be far more desirable to plant *Eucalyptus* plantations rather than those of the *Pinus* genus. Di Castri and Vitali - di Castri (1981) noted drastic changes in the

biological conditions of the soil in Chile under *Pinus radiata* plantations, in spite of their being adjacent to native formations and *Eucalyptus* plantations. Springett (1976b) and Di Castri and Vitali - di Castri (1981) noted the mediterranean soils were unable to cope with the high acidity of the pine needles, the impoverished soil microfauna of the pine stands were unable to decompose pine or sclerophyll litter as fast as the full complement of soil microfauna in the native vegetation. There are also advantages to planting mixed tree species; the production of mixed litter increases the heterogeneity of litter substrates, thus increasing resource and microhabitat diversity for the soil-litter communities (Chapman *et al.*, 1988). Briones and Ineson (1996) also suggest that mixtures of *Eucalyptus globulus* litter with other litters could maintain the high productivity rates of these plantations.

Conclusion

This study was the first known of its kind in Australia, and possibly the world, providing important baseline data on the soil and litter mite diversity in monoculture *Eucalyptus globulus* plantations. These introduced monocultures are now a prominent feature in the southwest region of Western Australia. It is clear that such plantations planted on pasturelands increase the soil biodiversity as a result of increased habitat complexity, broadened resource base and reduced climatic fluctuations. This was particularly true for the oribatid mites. However, biodiversity of these young managed plantations was well below that of the native jarrah forest, a consequence of a narrowed trophic niche with respect to reduced heterogeneity, and thus reduced trophic dimensions.

The connection between soil-litter mite biodiversity and the ecosystem process of decomposition have now been well established. Therefore, maintenance of a diverse fauna is essential for the optimum functioning of these processes. While plantations have enhanced biodiversity over the pasture, there ought to be management of biodiversity within these ecosystems. One important recommendation arising from this work is that, rather than planting a single monoculture, plant a mixed tree stratum.

CHAPTER 5: REFERENCES

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APPENDIX I

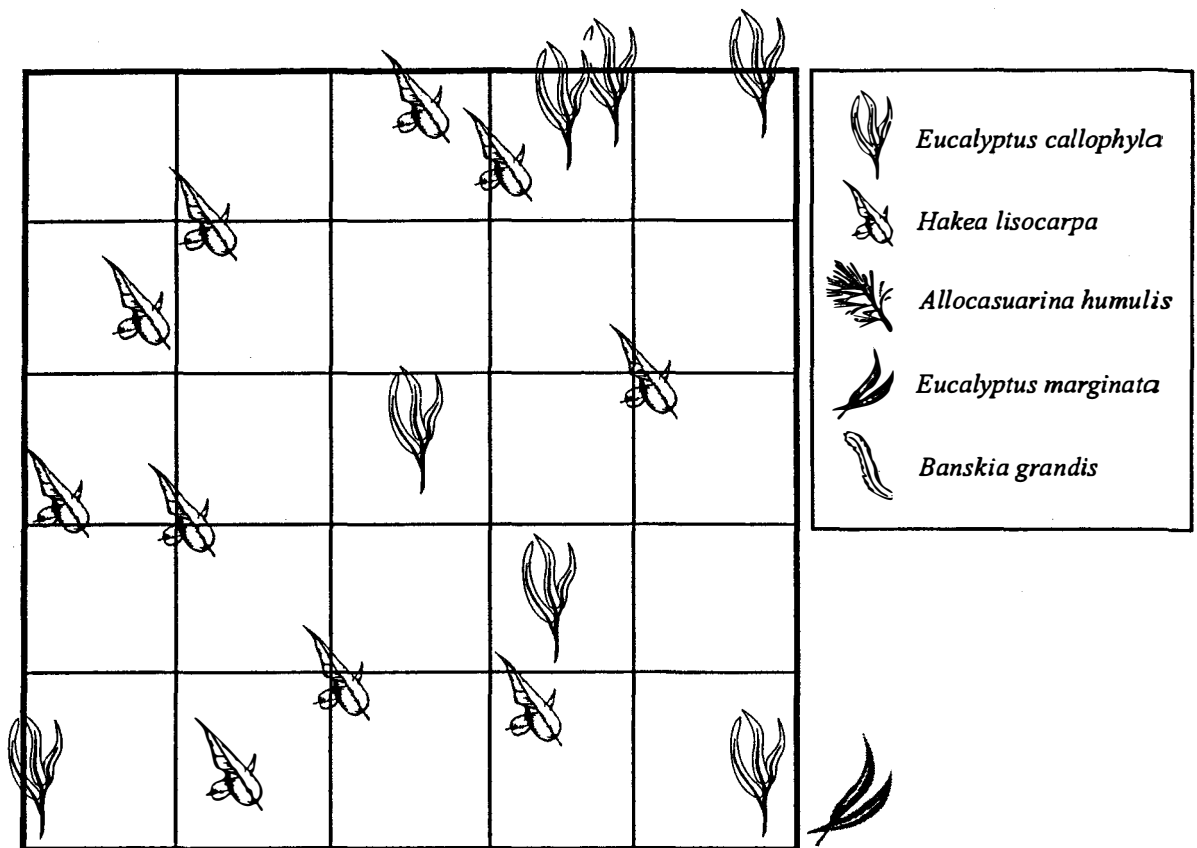


Figure A.I(a) Vegetation map within the 10m x 10m sampling quadrat of native forest spring site 1.

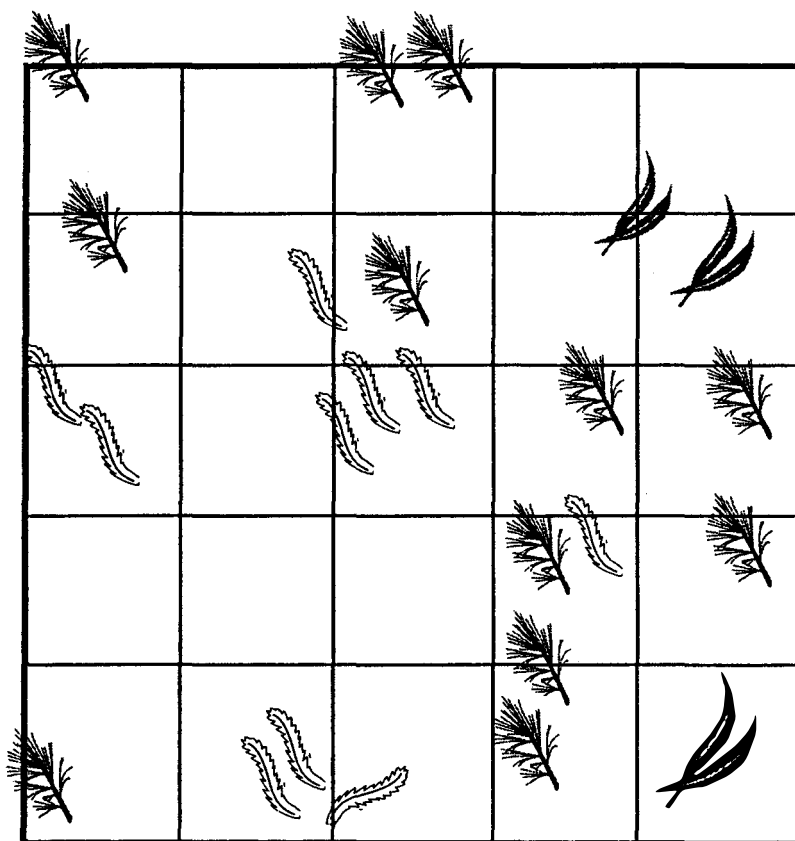


Figure A.I(b) Vegetation map within the 10m x 10m sampling quadrat of native forest spring site 2.

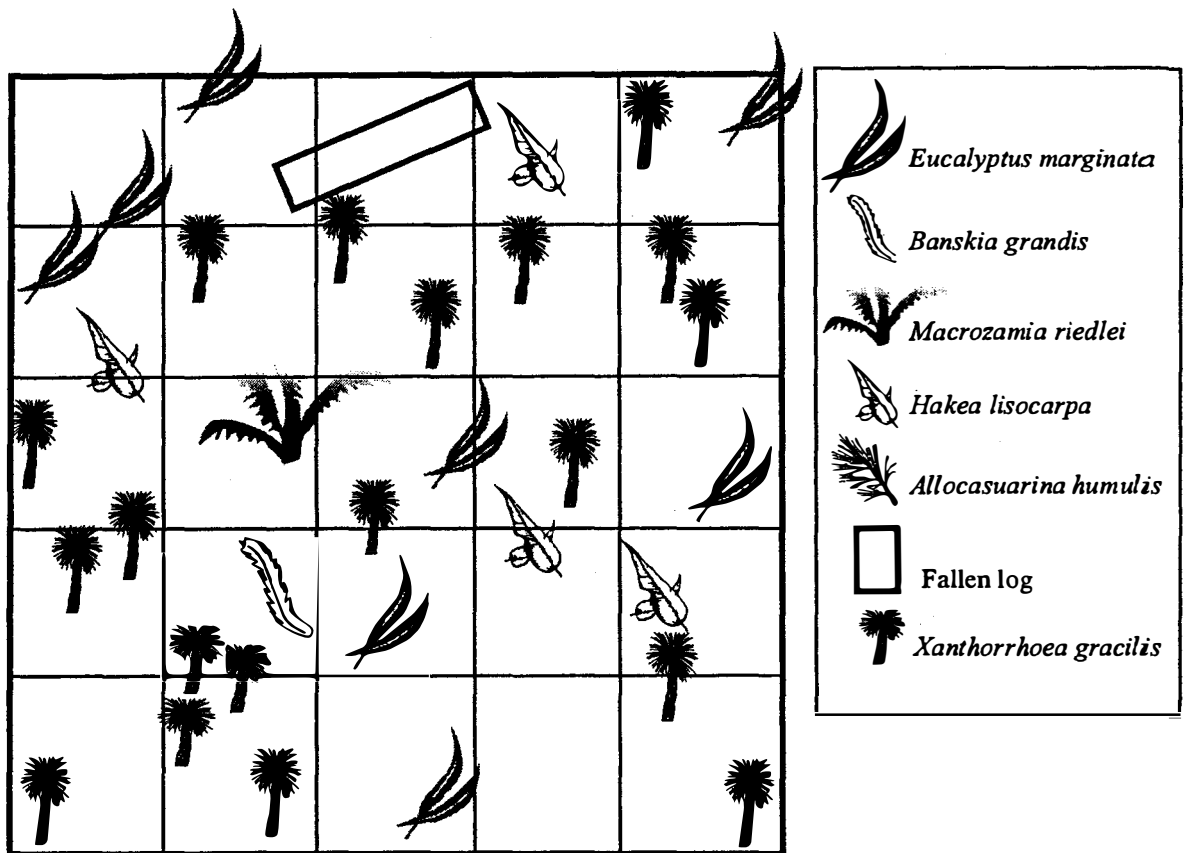


Figure A.I(c) Vegetation map within the 10m x 10m sampling quadrat of native forest spring site 3.

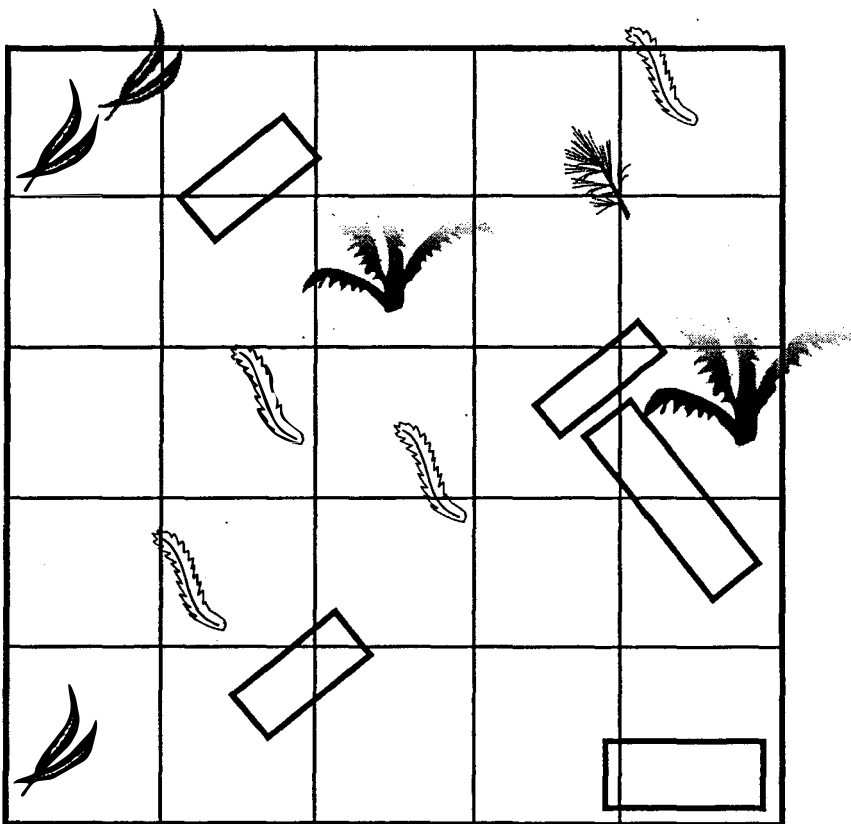


Figure A.I(d) Vegetation map within the 10m x 10m sampling quadrat of native forest autumn site 1.

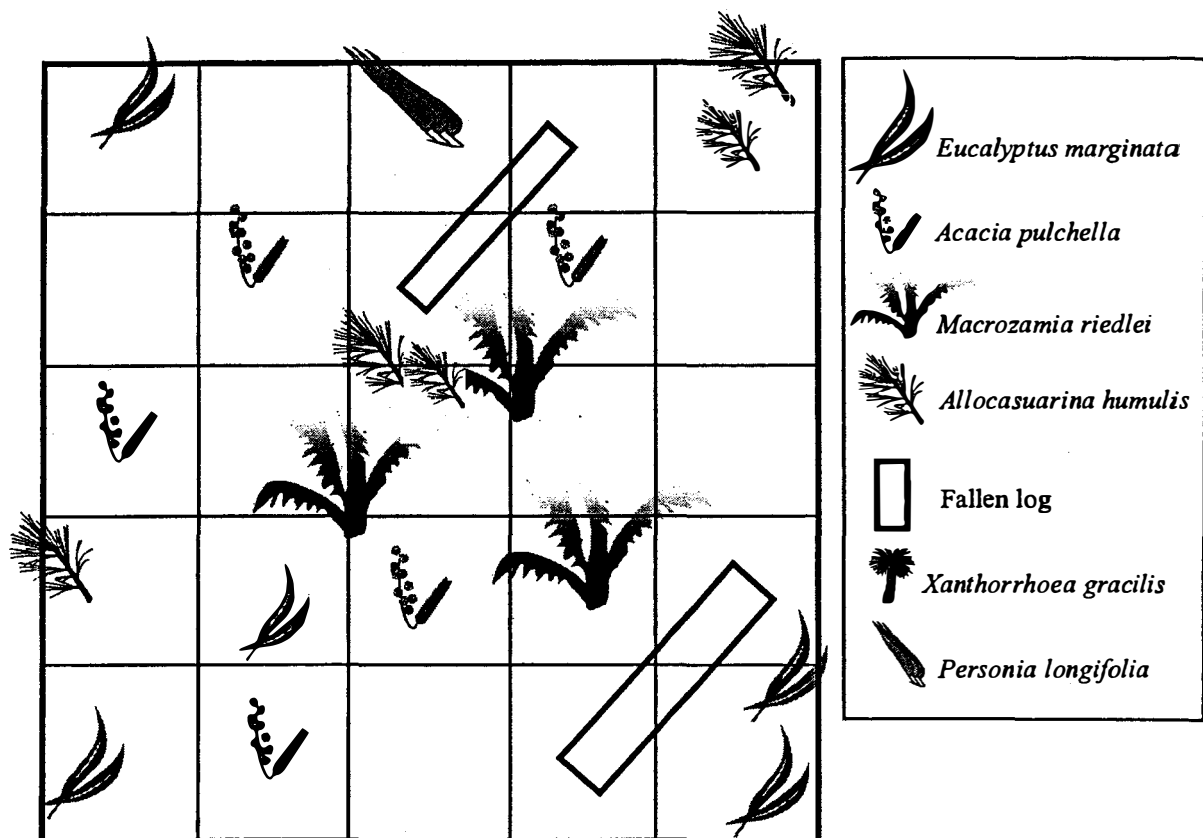


Figure A.I(e) Vegetation map within the 10m x 10m sampling quadrat of native forest autumn site 2.

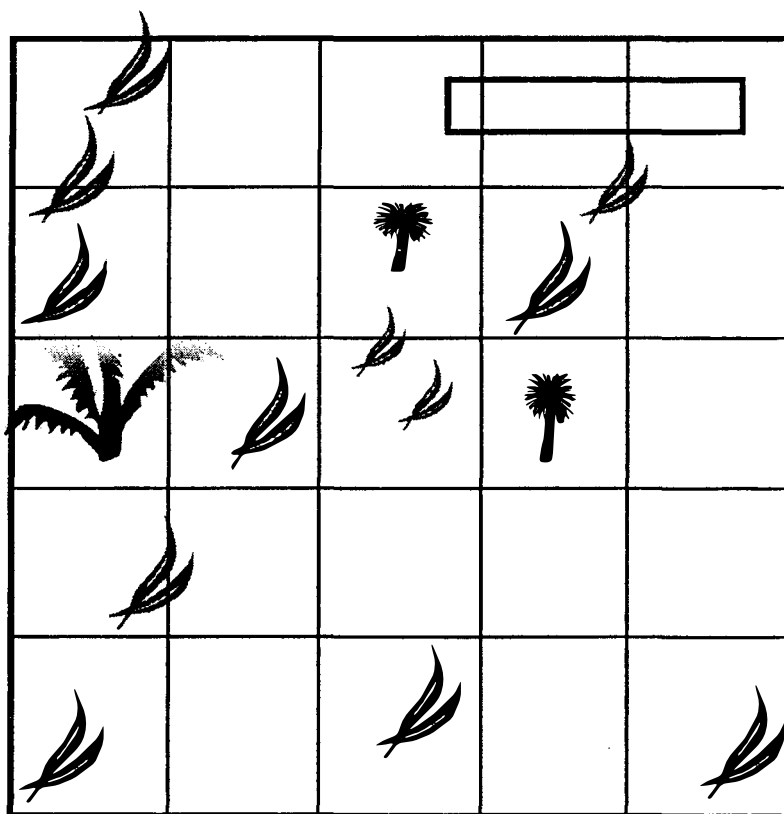


Figure A.I(f) Vegetation map within the 10m x 10m sampling quadrat of native forest autumn site 3.

APPENDIX II

Appendix II. Presence (+) and absence (-) of acarine species extracted from surface litter and soil during the sampling periods of spring 1997 and autumn 1998.

		<i>E. marginata</i> forest		<i>E. globulus</i> plantation		Pasture
		Litter	Soil	Litter	Soil	Soil
Astigmata						
	Family Acaridae					
	<i>Tyrophagus sp.1</i>	+	+	+	+	+
Cryptostigmata						
	Family Astegistidae					
	<i>Cultroribella sp.1</i>	-	+	-	-	-
	Family Brachychthoniidae					
	<i>Sp.1</i>	+	+	-	-	-
	<i>Sp.3</i>	+	+	-	-	-
	Family Ceratozetidae					
	<i>Sp.1</i>	+	+	-	-	-
	Family Cymbaeremaeoidea					
	<i>Scapheremaeus sp.1</i>	+	+	-	-	-
	Family Damacidae					
	<i>Sp.1</i>	-	+	-	-	-
	Family Eremaozetidae					
	<i>Eremaozetes sp.1</i>	+	+	-	-	-
	Family Galumnidae					
	<i>Sp.1</i>	+	+	-	-	-

Appendix II. continued

	<i>E. marginata</i> forest		<i>E. globulus</i> plantation		Pasture
	Litter	Soil	Litter	Soil	Soil
Family Haplochthoniidae					
<i>Haplochthonius</i> sp.1	-	+	-	+	-
<i>Haplochthonius</i> sp.2	-	+	-	-	-
Family Haplozetidae					
<i>Sp.1</i>	-	+	-	-	-
Family Humerotatidae					
<i>Humerotates</i> sp.1	+	-	-	-	-
Family Mycobatiidae					
<i>Antarctiozetes</i> sp.2	+	-	-	-	-
<i>Mycobates</i> sp.1	+	+	-	-	-
<i>Sp.1</i>	+	+	+	+	-
Family Oppiidae					
<i>Sp.1</i>	+	+	-	+	-
<i>Sp.2</i>	-	+	-	+	-
<i>Sp.3</i>	-	+	-	-	-
<i>Sp.4</i>	-	+	-	-	-
<i>Sp.5</i>	-	+	-	-	-
<i>Sp.6</i>	-	+	-	-	-
Subfamily Oppiinae					
<i>Sp.1</i>	+	+	+	+	-
<i>Sp.2</i>	+	+	-	+	-
<i>Sp.3</i>	+	+	+	+	-

Appendix II. continued

	<i>E. marginata</i> forest		<i>E. globulus</i> plantation		Pasture
	Litter	Soil	Litter	Soil	Soil
Subfamily Oppiellinae					
<i>Sp.1</i>	+	+	+	+	-
Family Oribatulidae					
<i>Sp.1</i>	+	-	-	-	-
<i>Sp.2</i>	+	+	-	-	-
<i>Sp.4</i>	+	+	-	-	-
<i>Sp.5</i>	-	+	-	-	-
<i>Sp.6</i>	-	+	-	+	-
Family Phenopelopidae					
<i>Sp.1</i>	+	-	-	-	-
Family Plateremaeidae					
<i>Sp.1</i>	+	-	-	-	-
Family Scheloribatidae					
<i>Scheloribates sp.1</i>	+	-	-	-	-
<i>Scheloribates sp.2</i>	+	-	-	-	-
<i>Sp.1</i>	+	-	-	-	-
<i>Sp.2</i>	+	-	-	-	-
Family Scutoverticidae					
<i>Scutovertex sp.1</i>	+	-	-	-	-
Family Spherochthoniidae					
<i>Sp.1</i>	+	+	-	-	-

Appendix II. continued

	<i>E. marginata</i> forest		<i>E. globulus</i> plantation		Pasture
	Litter	Soil	Litter	Soil	Soil
Family Tectocepheidae					
<i>Tectocepheus velatus</i>	+	+	+	-	-
<i>Tectocepheus sp.2</i>	+	-	-	-	-
<i>Tegeozetes sp.1</i>	+	+	+	+	-
<i>Sp.1</i>	-	+	-	-	-
Unknown <i>sp.1</i>	+	+	-	+	-
Unknown <i>sp.2</i>	-	+	-	-	-
Unknown <i>sp.3</i>	-	+	-	-	-
Mesostigmata					
Family Ascidae					
<i>Sp.1</i>	+	+	+	+	-
<i>Sp.2</i>	-	+	-	-	-
Superfamily Dermanyssoidea					
Unknown <i>sp.1</i>	+	+	-	+	+
Unknown <i>sp.2</i>	+	-	-	-	-
Family Eviphididae					
<i>Sp.1</i>	+	+	-	-	-
Family Laelapidae					
<i>Sp.1</i>	-	+	-	-	-
Family Phytoseiidae					
<i>Sp.1</i>	+	+	+	+	+

Appendix II. *continued*

	<i>E. marginata</i> forest		<i>E. globulus</i> plantation		Pasture
	Litter	Soil	Litter	Soil	Soil
Superfamily Rhodacaroidea					
Family Ologomasidae					
<i>Sp.1</i>	+	+	-	-	-
<i>Sp.2</i>	+	-	+	-	-
<i>Sp.3</i>	+	-	-	+	-
<i>Sp.5</i>	+	-	-	-	-
Superfamily Uropodoidea					
Unknown <i>sp.3</i>	+	-	-	-	-
Family Uropodidae					
<i>Sp.1</i>	+	+	-	-	-
<i>Sp.2</i>	-	+	-	-	-
Unknown <i>sp.1</i>	+	+	-	-	+
Unknown <i>sp.2</i>	+	-	-	-	-
Prostigmata					
Family Anystidae					
<i>Sp.1</i>	+	-	-	-	-
<i>Sp.2</i>	-	+	-	-	-
Family Bdellidae					
Subfamily Odontoscirinae					
<i>Odontoscirus sp.1</i>	+	+	+	-	+
<i>Odontoscirus sp.2</i>	-	+	-	+	-
<i>Odontoscirus sp.3</i>	+	-	+	-	-
Subfamily Spinibdellinae					
<i>Spinibdella sp.1</i>	+	+	+	+	-

Appendix II. continued

	<i>E. marginata</i> forest		<i>E. globulus</i> plantation		Pasture
	Litter	Soil	Litter	Soil	Soil
Family Caeculidae					
<i>Sp.1</i>	-	+	-	-	-
Family Camerobiidae					
<i>Sp.1</i>	-	-	-	+	-
Family Cheyletidae					
<i>Sp.1</i>	+	-	+	-	-
Family Cryptognathidae					
<i>Sp.1</i>	+	-	-	-	-
Family Cunaxidae					
<i>Sp.1</i>	-	-	+	-	-
<i>Sp.2</i>	-	-	+	-	-
<i>Sp.3</i>	-	-	-	-	+
<i>Cunaxa sp.1</i>	-	-	+	-	-
Family Erythraeidae					
<i>Sp.1</i>	+	-	-	-	-
<i>Sp.2</i>	-	-	-	+	-
<i>Sp.3</i>	-	-	-	-	+
Order Eupodoidea					
Unknown <i>sp.1</i>	-	-	+	+	-
Unknown <i>sp.2</i>	-	+	-	-	-
Unknown <i>sp.3</i>	-	+	-	-	-

Appendix II. continued

	<i>E. marginata</i> forest		<i>E. globulus</i> plantation		Pasture
	Litter	Soil	Litter	Soil	Soil
Family Eupodidae					
<i>Sp.1</i>	+	-	-	-	-
<i>Sp.2</i>	+	+	-	-	-
<i>Sp.4</i>	-	-	-	+	-
<i>Sp.5</i>	-	-	-	+	-
Family Linotetranidae					
<i>Linotetrans sp.1</i>	-	+	-	-	-
Family Nanorchesidae					
<i>Sp.1</i>	+	+	-	-	-
Family Pachygnathidae					
<i>Sp.1</i>	+	+	-	-	-
Family Penthelodidae					
<i>Sp.1</i>	-	+	-	-	-
<i>Sp.2</i>	-	+	-	-	+
<i>Sp.3</i>	-	+	-	-	-
Family Pseudocheylidae					
<i>Anoplocheylus sp.1</i>	+	+	+	-	-
<i>Sp.3</i>	-	+	-	-	-
<i>Sp.4</i>	-	+	-	-	-

Appendix II. *continued*

	<i>E. marginata</i> forest		<i>E. globulus</i> plantation		Pasture
	Litter	Soil	Litter	Soil	Soil
Family Raphignathidae					
<i>Sp.1</i>	+	-	-	-	-
<i>Sp.2</i>	-	+	-	+	-
Family Rhagidiidae					
<i>Sp.1</i>	+	-	-	-	-
<i>Sp.2</i>	-	-	+	+	-
<i>Sp.4</i>	-	-	-	+	-
<i>Sp.5</i>	-	-	-	+	+
<i>Sp.6</i>	-	-	-	+	-
Family Smarididae					
<i>Sp.1</i>	+	-	-	-	-
<i>Sp.2</i>	+	-	-	-	-
Family Stigmaeidae					
<i>Sp.1</i>	+	-	+	+	-
<i>Sp.2</i>	-	+	+	-	-
<i>Sp.3</i>	-	-	-	+	-
<i>Sp.4</i>	-	-	-	+	-
Family Tarsonemidae					
<i>Sp.1</i>	+	+	+	-	-
Family Tetranychidae					
<i>Sp.1</i>	+	-	+	+	-
<i>Sp.2</i>	-	+	-	-	+

Appendix II. continued

	<i>E. marginata</i> forest		<i>E. globulus</i> plantation		Pasture
	Litter	Soil	Litter	Soil	Soil
Unknown <i>sp. 1</i>	-	+	-	-	-
Unknown <i>sp.2</i>	-	-	-	+	-

APPENDIX III

APPENDIX III. The Impact of Vegetation Changes on the Collembola

Although not the main focus of the present study, the springtails deserve to be discussed because they are major components of the mesofauna and have a similar ecological niche to the Oribatida.

Springtail abundances were consistently higher in the modified ecosystems compared to the native jarrah forest with abundances increasing along the gradient from native forest to *E. globulus* plantation to grazed pasture. Di Castri and Vitali - di Castri (1981) also found springtail abundances increased from the sclerophyll forest to the *E. globulus* plantation to the irrigated pasture, in which soil abundances under the pasture were significantly greater than in the other two ecosystems. In contrast to these findings, Pinto *et al.* (1997) found lower abundances of springtails in the *E. globulus* plantation soils, and related this to differences in the chemical composition of leaves, such as corresponding nitrogen contents. Serralheiro and Madeira (1990) found springtail population densities to be significantly higher under the native vegetation than the *E. globulus* plantation in the soil and forest floor. Moreover, Pinto *et al.* (1997) illustrated that springtail species richness and diversity was higher in native forests than in introduced *E. globulus* plantations.

The greater number of springtails extracted from the modified ecosystems may be related to the phosphorous content of the remaining groundcover. It has been suggested that an increase in the number of species, particularly introduced species, on improved pastures may represent a response to dietary quality (King, Greenslade and Hutchinson, 1985)

associated with fertilizer applications. To take advantage of the enhanced nutritional opportunity, introduced species would appear to have the ability to use resources quickly when they become available (King *et al.*, 1985). The higher numbers in the modified ecosystems of this study may be the result of an increase in the number of introduced species. Interestingly, the effect of sheep stocking intensity did not appear to reduce springtail abundances in the grazed pasture despite being a strong influence in other studies (King and Hutchinson, 1976; Hutchinson and King, 1980; King and Hutchinson, 1983). Increased sheep numbers (hence increased grazing intensity) decreased springtail proportions and species richness in the study of King *et al.* (1985). The increased abundances in the *E. globulus* plantation of the present study could also be related to the community being indirectly attracted to roots that are colonized by a diversity of microfungi (Klironomos and Kendrick, 1995), particularly if the seedlings were inoculated prior to planting. Springtails can be highly specialized feeders on plant residues and fungi, with floristic richness of vascular plants and microfungi a determinant of their extent.