QUANTIFYING STAND STRUCTURAL COMPLEXITY IN WOODLAND AND DRY SCLEROPHYLL FOREST, SOUTH-EASTERN AUSTRALIA

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A thesis submitted for the degree of Doctor of Philosophy of the Australian National University
STATEMENT OF ORIGINALITY

I declare that this thesis represents an original piece of work, and does not contain, in part or in full, the published work of any other individual, except where acknowledged.

Chris McElhinny,
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# Table of contents

List of Tables ...................................................................................................................... i

List of Figures .................................................................................................................... iii

Abstract ................................................................................................................................. v

Chapter 1: Introduction ....................................................................................................... 1

Thesis overview ................................................................................................................... 1

1.1 Biodiversity .................................................................................................................... 2

1.1.1 Defining biodiversity ................................................................................................. 2

1.1.2 Biodiversity indicators .............................................................................................. 3

1.2 Research questions ......................................................................................................... 6

Chapter 2: Stand structure and structural attributes ......................................................... 7

Chapter summary .................................................................................................................. 7

2.1 Defining forest and woodland structure ....................................................................... 7

2.1.1 Ecosystem structure, function and composition ......................................................... 7

2.1.2 Structural attributes .................................................................................................. 8

2.1.3 Stand structure ......................................................................................................... 8

2.2 Stand structural attributes ............................................................................................ 9

2.2.1 Overview of attributes ............................................................................................... 9

2.2.2 Foliage ....................................................................................................................... 13

2.2.3 Canopy cover ............................................................................................................ 15

2.2.4 Tree diameter ........................................................................................................... 16

2.2.5 Tree height .............................................................................................................. 20

2.2.6 Tree spacing ............................................................................................................ 21

2.2.7 Tree species ............................................................................................................ 23

2.2.8 Understorey vegetation ............................................................................................. 23

2.2.9 Dead wood .............................................................................................................. 25

2.3 Conclusions .................................................................................................................. 28

Chapter 3: Indices of stand structural complexity ......................................................... 29

Chapter summary .................................................................................................................. 29

3.1 Stand level indices of structural complexity ............................................................... 29

3.1.1 Overview of indices ................................................................................................. 29
Chapter 4: Fauna-habitat relationships, a basis for identifying key structural attributes in temperate Australian forests and woodlands ..... 44

Chapter summary .............................................................................................................. 44
4.1 Introduction .............................................................................................................. 44
4.2 The type of studies reviewed .................................................................................. 45
4.3 Habitat resources and structural attributes for seven faunal groups ........ 46
   4.3.1 Habitat resources used by birds .................................................................. 46
   4.3.2 Habitat resources used by arboreal mammals .......................................... 52
   4.3.3 Habitat resources used by ground mammals ............................................. 58
   4.3.4 Habitat resources used by bats .................................................................. 61
   4.3.5 Habitat resources used by reptiles ............................................................... 64
   4.3.6 Habitat resources used by amphibians ....................................................... 68
   4.3.7 Habitat resources used by invertebrates ..................................................... 69
4.4 Synthesis: key structural attributes ...................................................................... 74
   4.4.1 Generic attributes ....................................................................................... 74

Chapter 5: A method for collecting structural attribute data in a representative sample of stands ................................................................. 78

Chapter summary .............................................................................................................. 78
5.1 Introduction .............................................................................................................. 78
5.2 Study area ............................................................................................................... 79
5.3 Data collection ........................................................................................................... 81
   5.2.1 Sampling design .......................................................................................... 81
   5.2.2 Measurements taken .................................................................................... 86

Chapter 6: Arriving at a core set of attributes for characterising structural complexity in dry sclerophyll forests and woodlands of south-eastern Australia ................................................................. 91

Chapter summary .............................................................................................................. 91
Chapter 7: An index for rapidly assessing stand structural complexity in
dry sclerophyll forests and woodlands of south-eastern Australia .......121

Chapter summary........................................................................................................121

7.1 Introduction .............................................................................................................121

7.2 Combining attributes in an index.........................................................................122

7.2.1 Index framework.................................................................................................122

7.2.2 Rescaling attributes.........................................................................................123

7.2.3 Prototype Index..................................................................................................127

7.3 Weighting of attributes in the index ....................................................................130

7.4 Applying the index in a user-friendly spreadsheet.............................................134

7.4.1 Data entry ........................................................................................................134

7.4.2 Calculating an index score and site ranking....................................................137

7.4.3 Calculating an index score using a subset of attributes ..................................137

7.4.4 Outputs for individual attributes.......................................................................139
Chapter 8: Comparing the performance of the index to other indices used in temperate Australian ecosystems .......................................................... 141

Chapter summary .......................................................................................... 141

8.1 Introduction .............................................................................................. 142

8.2 Indices which score attributes on an absolute scale ............................... 143

8.2.1 Habitat Complexity Score (Newsome and Catling, 1979) ...................... 143

8.2.2 Habitat Complexity Score (Watson et al., 2001) .................................. 146

8.3 Indices which score attributes relative to benchmarks ......................... 147

8.3.1 Site Condition Score - Habitat Hectares Index .................................. 153

8.3.2 Vegetation Condition Score – Biodiversity Benefits Index .................. 158

8.3.3 Vegetation Condition Score – BioMetric Assessment Tool ................. 162

8.4 Comparing the discrimination power of the different indices ................. 167

Chapter 9: Conclusions ................................................................................ 170

9.1 Introduction .............................................................................................. 170

9.2 Defining stand structural complexity ....................................................... 170

9.3 Identifying, scoring and combining structural attributes in an index ...... 171

9.3.1 Identifying attributes ......................................................................... 171

9.3.2 Scoring attributes ............................................................................. 172

9.3.3 Combining and weighting attributes .................................................. 173

9.4 The practical application of the index ..................................................... 174

9.5 Applying the index to other vegetation types .......................................... 175

9.6 Further research ..................................................................................... 176

9.6.1 Thresholds ....................................................................................... 176

9.6.2 Remote sensing ................................................................................ 177

References .................................................................................................... 178

Appendix 1 ..................................................................................................... 201

Appendix 2 ..................................................................................................... 206
List of Tables

Table 1: Attributes used to characterise stand structure...........................................10
Table 2: Indices used to quantify stand structural complexity........................................31
Table 3: The percentage of 55 studies that addressed seven broad faunal groups......45
Table 4: Structural attributes significantly associated with the presence, abundance or richness of one or more bird species in eucalypt forests or woodlands............47
Table 5: Structural attributes significantly associated with the presence, abundance or richness of arboreal mammal species in eucalypt forests or woodlands.................54
Table 6: Structural attributes significantly associated with the presence, abundance or richness of ground mammal species in eucalypt forests or woodlands...........59
Table 7: Structural attributes significantly associated with the presence, abundance or richness of bats in eucalypt forests or woodlands.........................................................62
Table 8: Structural attributes significantly associated with the presence, abundance or richness of reptiles in eucalypt forests or woodlands..................................................65
Table 9: Structural attributes significantly associated with the abundance or richness of amphibians in eucalypt forests or woodlands.........................................................69
Table 10: Structural attributes significantly associated with the presence, abundance or richness of invertebrates in eucalypt forests or woodlands........................................71
Table 11: Attributes significantly associated with the presence or abundance of major faunal groupings in temperate eucalypt forests or woodlands..............................75
Table 12: An operational set of stand level attributes, for the collection of field data...77
Table 13: The three vegetation communities selected for this study.........................81
Table 14: 48 study sites across 24 strata........................................................................84
Table 15: Summary of measurements taken at each plot. Measurements are grouped under the structural element they address and the sampling method used to collect the data........................................................................................................88
Table 16: Mean values and ranges for a selection of structural attributes in one woodland and two dry sclerophyll forest communities.............................................93
Table 17: Thirteen core structural attributes identified from an initial suite of 75........96
Table 18: Mean projective foliage cover of perennial vegetation within different lifeforms and height classes. .................................................................97
Table 19: Scoring a Hollow Diversity Index on the basis of hollow-bearing tree abundance in three hollow size classes.................................................................106
Table 20: Scoring an index of coarse woody debris diversity on the basis of decay class volume ........................................................................................................111
Table 21: Percentage of 48 study sites occurring in each of the four litter cover classes
suggested by Newsome and Catling (1979), and Watson et al. (2001), as a component of Habitat Complexity Score

Table 22: Regression equations for scoring attributes on a scale of 0-10 as a function of the raw attribute data

Table 23: Attribute scores on a scale from 0-10 were highly correlated with the original attribute data

Table 24: Habitat Complexity Score of Newsome and Catling (1979)

Table 25. Percentage of 48 study sites contained in each of the four canopy cover classes suggested by Newsome and Catling (1979), and Watson et al. (2001).

Table 26: Habitat Complexity Score of Watson et al. (2001)

Table 27. Comparison between the 13 attributes included in my proposed index and attributes included in the stand scale component of three indices currently used to assess vegetation condition in temperate Australian ecosystems, and which score attributes relative to benchmarks.

Table 28: Comparison between benchmarks established using data from 16 high quality dry sclerophyll forest sites, and 8 high quality woodland sites, and benchmarks proposed for comparable ecological vegetation communities in Victoria on the basis of expert opinion.

Table 29: Benchmarks and class intervals, used to score 16 woodland and 32 dry sclerophyll forest study sites according to the Site Condition Score component of the Habitat Hectares Index of Parkes et al. (2003)

Table 30: Benchmarks and class intervals, used to score 16 woodland and 32 dry sclerophyll forest study sites according to the Vegetation Condition Score component of the Biodiversity Benefits Index of Oliver and Parkes (2003)

Table 31: Benchmarks and class intervals, used to score 16 woodland and 32 dry sclerophyll forest study sites, according to the Vegetation Condition Score component of the BioMetric Assessment Tool of Gibbons et al. (2004)
List of Figures

Figure 1: Structural complexity is a relative, rather than absolute concept; illustrated by three structurally complex stands from vegetation communities in southern Australia ...........................................................................................................41

Figure 2: Flow chart summarising the five-stage methodology proposed for the development of an index of structural complexity .........................................................43

Figure 3: The study area ..................................................................................................................80

Figure 4: At each site 3 plots were located along a stand transect running diagonally across the direction of the main environmental gradient and passing through the centre of the target stand ..................................................................................................85

Figure 5: Measurements taken at each plot ......................................................................................86

Figure 6. Most hollow-bearing trees occurred in the smaller diameter classes ..........104

Figure 7. Larger trees were more likely to contain hollows than smaller trees .................105

Figure 8. Litter dry weight distinguished between sites with similarly high levels of litter cover ........................................................................................................................................114

Figure 9. Using Principal Components Analysis the original axes or variables in a multidimensional space can be represented as vectors in a two dimensional space defined by the first two principal components .................................................................117

Figure 10. The first two principal components (PC1 and PC2) explained a total of 59% of the variation between study sites in the ecological space mapped by the 13 core attributes ..............................................................................................................................118

Figure 11. Attributes were rescaled as a score from 0-10 by fitting linear regressions to the quartile midpoints of the attribute distributions in woodland and dry sclerophyll sites .......................................................................................................................125

Figure 12. The prototype index differentiated between study sites, with index values approximately normally distributed in both woodland and dry sclerophyll sites 129

Figure 13. The prototype index ranked sites in a logical way in woodland and dry sclerophyll forest .................................................................................................................................130

Figure 14. One of 10,000 simulations used to quantify the correlation between unweighted and weighted index scores for 48 woodland and dry sclerophyll forest sites .................................................................................................................................132

Figure 15 Distribution of the correlation (Pearson’s r) between weighted and unweighted index scores in 48 study sites, for 10,000 random weightings of index attributes .................................................................................................................................133

Figure 16. Summary of data requirements, and outputs produced, by a user-friendly spreadsheet incorporating the final version of the index .................................................................................135
Figure 26: The use of broad classes to score attributes in the Vegetation Condition Score (VCS).

Figure 24: The Vegetation Condition Score (VCS).

Figure 22: The use of broad classes to score attributes in the Site Condition Score of Parkes et al. (2003) masked differences between sites and awarded most sites with high scores.

Figure 23: The Site Condition Score (SCS) component of the Habitat Hectares Index of Parkes et al. (2003) had an approximately normal distribution when applied to 32 dry sclerophyll forest study sites and 16 woodland study sites.

Figure 24: The Vegetation Condition Score (VCS) component of the Biodiversity Benefits Index of Oliver and Parkes (2003) had an approximately normal distribution when applied to 32 dry sclerophyll forest study sites and 16 woodland study sites.

Figure 26: The use of broad classes to score attributes in the Vegetation Condition Score of Gibbons et al. (2004) masked differences between sites and awarded most sites with high scores.

Figure 27: The Vegetation Condition Score (VCS) component of the BioMetric Assessment Tool of Gibbons et al. (2004) had a distribution skewed towards high scores when applied to 32 dry sclerophyll forest study sites and 16 woodland study sites.

Figure 28: There were clear differences between the discrimination power of my proposed index and five other indices.
Abstract

In this thesis I present and test a methodology for developing a stand scale index of structural complexity. If properly designed such an index can act as a summary variable for a larger set of stand structural attributes, providing a means of ranking stands in terms of their structural complexity, and by association, their biodiversity and vegetation condition. This type of index can also facilitate the use of alternative policy instruments for biodiversity conservation, such as mitigation banking, auctions and offsets, that rely on a common currency – the index value – that can be compared or traded between sites. My intention was to establish a clear and documentable methodology for developing a stand scale index of structural complexity, and to test this methodology using data from real stands.

As a starting point, I reviewed the literature concerning forest and woodland structure and found there was no clear definition of stand structural complexity, or definitive suite of structural attributes for characterising it. To address this issue, I defined stand structural complexity as a combined measure of the number of different structural attributes present in a stand, and the relative abundance of each of these attributes. This was analogous to approaches that have quantified diversity in terms of the abundance and richness of elements. It was also concluded from the review, that stand structural complexity should be viewed as a relative, rather than absolute concept, because the potential levels of different structural attributes are bound within certain limits determined by the inherent characteristics of the site in question, and the biota of the particular community will have evolved to reflect this range of variation. This implied that vegetation communities with naturally simple structures should have the potential to achieve high scores on an index of structural complexity.

I proposed the following five-stage methodology for developing an index of stand structural complexity:

1. Establish a comprehensive suite of stand structural attributes as a starting point for developing the index, by reviewing studies in which there is an established relationship between elements of biodiversity and structural
attributes.

2. Develop a measurement system for quantifying the different attributes included in the comprehensive suite.

3. Use this measurement system to collect data from a representative set of stands across the range of vegetation condition (highly modified to unmodified) and developmental stages (regrowth to oldgrowth) occurring in the vegetation communities in which the index is intended to operate.

4. Identify a core set of structural attributes from an analysis of these data.

5. Combine the core attributes in a simple additive index, in which attributes are scored relative to their observed levels in each vegetation community.

Stage one of this methodology was addressed by reviewing a representative sample of the literature concerning fauna-habitat relationships in temperate Australian forests and woodlands. This review identified fifty-five studies in south-east and south-west Australia, in which the presence or abundance of different fauna were significantly (p<0.05) associated with vegetation structural attributes. The majority of these studies concerned bird, arboreal mammal, and ground mammal habitat requirements, with relatively fewer studies addressing the habitat requirements of reptiles, invertebrates, bats or amphibians. Thirty-four key structural attributes were identified from these fifty-five studies, by grouping similar attributes, and then representing each group with a single generic attribute. This set, in combination with structural attributes identified in the earlier review, provided the basis for developing an operational set of stand level attributes for the collection of data from study sites.

To address stages two and three of the methodology, data was collected from one woodland community –Yellow Box-Red Gum (E. melliodora-E. Blakelyi) – and two dry sclerophyll forest communities – Broadleaved Peppermint-Brittle Gum (E. dives-E. mannifera), Scribbly Gum-Red Stringybark (E. rossii-E. macrorhyncha) – in a 15,000 km² study area in the South-eastern Highlands Bioregion of Australia. A representative set of 48 sites was established within this study area, by identifying 24 strata, on the basis of the three vegetation communities, two catchments, two levels of rainfall and two levels of condition, and then locating two sites (replicates) within each stratum. At each site, three
plots were systematically established, to provide an unbiased estimate of stand level means for 75 different structural attributes.

I applied a three-stage analysis to identify a core set of attributes from these data. The first stage – a preliminary analysis – indicated that the 48 study sites represented a broad range of condition, and that the two dry sclerophyll communities could be treated as a single community, which was structurally distinct from the woodland community. In the second stage of the analysis, thirteen core attributes were identified using the criteria that a core attribute should:

- Be either, evenly or approximately normally distributed amongst study sites;
- Distinguish between woodland and dry sclerophyll communities;
- Function as a surrogate for other attributes;
- Be efficient to measure in the field.

The core attributes were: Vegetation cover <0.5m; Vegetation cover 0.5-6.0m; Perennial species richness; Lifeform richness; Stand basal area of live trees; Quadratic mean diameter of live stems; ln(number of regenerating stems ha\(^{-1}\)+1); ln(number of hollow bearing trees ha\(^{-1}\)+1); ln(number of dead trees ha\(^{-1}\)+1); \(\sqrt{\text{number of live stems ha}^{-1} > 40\text{ cm dbh}}\); \(\sqrt{\text{total log length ha}^{-1}}\); \(\sqrt{\text{total large log length ha}^{-1}}\); Litter dry weight ha\(^{-1}\). This analysis also demonstrated that the thirteen core attributes could be modelled as continuous variables, and that these variables were indicative of the scale at which the different attributes operated.

In the third and final stage of the analysis, Principal Components Analysis was used to test for redundancy amongst the core attributes. Although this analysis highlighted six groupings, within which attributes were correlated to some degree, these relationships were not considered sufficiently robust to justify reducing the number of core attributes.

The thirteen core attributes were combined in a simple additive index, in which, each attribute accounted for 10 points in a total index value of 130. Attributes were rescaled as a score from 0-10, using equations that modelled attribute
score as a function of the raw attribute data. This maintained a high correlation 
(r > 0.97, p < 0.0001) between attribute scores and the original attribute data. 
Sensitivity analysis indicated that the index was not sensitive to attribute 
weightings, and on this basis attributes carried equal weight. In this form my 
index was straightforward to apply, and approximately normally distributed 
amongst study sites.

I demonstrated the practical application of the index in a user-friendly 
spreadsheet, designed to allow landowners and managers to assess the 
condition of their vegetation, and to identify management options. This 
spreadsheet calculated an index score from field data, and then used this score 
to rank the site relative to a set of reference sites. This added a regional context 
to the operation of the index, and is a potentially useful tool for identifying sites 
of high conservation value, or for identifying sites where management actions 
have maintained vegetation quality. The spreadsheet also incorporated the 
option of calculating an index score using a subset of attributes, and provided a 
measure of the uncertainty associated with this score.

I compared the proposed index with five prominent indices used to quantify 
vegetation condition or habitat value in temperate Australian ecosystems. 
These were: Newsome and Catling’s (1979) Habitat Complexity Score, Watson 
et al.’s (2001) Habitat Complexity Score, the Site Condition Score component of 
the Habitat Hectares Index of Parkes et al. (2003), the Vegetation Condition 
Score component of the Biodiversity Benefits Index of Oliver and Parkes (2003), 
and the Vegetation Condition Score component of the BioMetric Assessment 
Tool of Gibbons et al. (2004). I found that my index differentiated between study 
sites better than each of these indices. However, resource and time constraints 
precluded the use of a new and independent data set for this testing, so that the 
superior performance of my index must be interpreted cautiously.

As a group, the five indices I tested contained attributes describing 
compositional diversity, coarse woody debris, regeneration, large trees and 
hollow trees – these were attributes that I also identified as core ones. However, 
unlike these indices, I quantified weeds indirectly through their effect on
indigenous plant diversity, I included the contribution of non-indigenous species to vegetation cover and did not apply a discount to this contribution, I limited the direct assessment of regeneration to long-lived overstorey species, I used stand basal area as a surrogate for canopy cover, I quantified litter in terms of biomass (dry weight) rather than cover, and I included the additional attributes of quadratic mean diameter and the number of dead trees.

I also concluded that Parkes et al. (2003), Oliver and Parkes (2003), and Gibbons et al. (2004), misapplied the concept of benchmarking, by characterising attributes in terms of a benchmark range or average level. This ignored processes that underpin variation at the stand level, such as the increased development of some attributes at particular successional stages, and the fact that attributes can respond differently to disturbance agents. It also produced indices that were not particularly sensitive to the differences in attribute levels occurring between stands. I suggested that a more appropriate application of benchmarking would be at the overarching level of stand structural complexity, using a metric such as the index developed in this thesis. These benchmarks could reflect observed levels of structural complexity in unmodified natural stands at different successional stages, or thresholds for structural complexity at which a wide range of biota are present, and would define useful goals for guiding on-ground management.
Chapter 1: Introduction

Thesis overview

In this thesis I present and test a methodology for developing a stand scale index of structural complexity. This methodology is applied to woodland and dry sclerophyll communities in the South-eastern Highlands Bioregion of Australia. The resulting index provides a quantitative tool for ranking stands from these communities in terms of their structural complexity, and by association, their biodiversity and vegetation condition. I also demonstrate how such an index can be used to provide practical, on ground advice for landowners and managers.

The thesis is structured in the following way:

- Chapter 1 is a brief overview of biodiversity indicators and the different scales at which they operate. This introduction provides a context for stand structural complexity as a fine scale indicator of biodiversity, which can inform on-ground management decisions.
- In Chapter 2 stand structure is defined in terms of structural attributes and stand structural complexity, and a range of stand level structural attributes used to characterise stand structure is identified.
- Chapter 3 presents a critique of existing indices of stand structural complexity, and this analysis provides a basis for the thesis methodology, which is presented at the end of the chapter.
- Chapter 4 addresses the first stage of my methodology by identifying a comprehensive suite of structural attributes through a review of fauna-habitat studies in temperate Australian forests and woodlands.
- Chapter 5 details the method used to quantify these structural attributes, and to collect data from 144 plots in 48 sites, located within a 15,000 km$^2$ study area.
- Chapter 6 presents the results of a three-stage analysis of these data, in which 13 key structural attributes are identified from an initial set of 104.
- In Chapter 7 these 13 key attributes are combined in an index of structural complexity. This index is presented as a user-friendly spreadsheet designed to allow landowners and managers to assess the condition of their vegetation and identify management options.
- Chapter 8 compares the performance of the proposed index to other indices currently used to quantify fauna habitat or vegetation condition in temperate Australian forest ecosystems.
- Chapter 9 is a summary and discussion of the key results of the study and indicates how these could be applied to other vegetation communities.
1.1 Biodiversity

1.1.1 Defining biodiversity

The starting point for this thesis is the concept of biodiversity. It is a relatively recent term, attributed to Walter Rosen in 1986 (Takacs, 1996), and is typically defined as the variety of life from genes to kingdoms and the diversity of interactions and processes at these many levels of organisation (Sarkar and Margules, 2002). Scale is an implied and key component of this definition, because the interactions and processes underpinning biodiversity can vary according to whether the scale is national (tens of millions km$^2$), regional (1000 ha to millions of km$^2$) landscape (100-1000 ha) or patch (1-100 ha) (Williams, 2004).

Despite its recent pedigree many governments are now committed to the maintenance of biodiversity through a range of initiatives, including the sustainable management of forest and woodland ecosystems. This largely reflects a range of national (e.g. Australian State of the Environment Committee, 2001; Australian and New Zealand Environment Conservation Council, 2000) and international processes (e.g. Montreal Process Working Group, 1995) established following the United Nations Convention on Biological Diversity, held in Rio de Janeiro 1992 (Grayson and Maynard, 1997).

The all-encompassing nature of biodiversity makes it an important concept, but also an extremely nebulous one. For example, Williams (2004) indicates that biodiversity is too a broad a concept to measure in its entirety, and that it has a range of different values dependent on the context in which it is being used. If land managers, planners and policy makers are to fulfil any of the commitments made to the maintenance of biodiversity they therefore need unambiguous and practical ways of defining and measuring this concept (Williams, 2004; Sarkar and Margules, 2002). This will require biodiversity to be redefined in terms of measurable attributes relevant to the scale and purpose for which it is being assessed (Williams, 2004).

On the basis that biodiversity equates to a diversity of life forms (Hunter, 1999),
two common approaches have been to describe biodiversity in terms of either species richness, or species diversity (Sarkar and Margules, 2002; Claridge et al., 1997) – for which a plethora of different indices have been proposed (e.g. Magurran, 1988). In practice these approaches have proved problematic because of the variety of different methods used, their lack of agreement with each other (Hurlbert, 1971), the impossibility of recording all species (Sarkar and Margules, 2002; Van Den Meersschaut and Vandekerkhove, 1998), and the fact that recording the presence and abundance of different species does not necessarily provide insight as to how to improve management (Stone and Porter, 1998). Richness is also unlikely to adequately capture difference and variety at national or regional levels. This is because it prioritises communities with high richness, such as tropical rainforests, above those with inherently low richness, such as woodlands and dry forests (Margules and Pressey, 2000).

1.1.2 Biodiversity indicators

An alternative approach to attempting a complete species census, is to use indicators, or measurable surrogates, to describe and monitor biodiversity (Sarkar and Margules, 2002; Ferris and Humphrey, 1999; Noss, 1990). At a regional and landscape scale, a common approach has been to use physical and climatic variables to classify different environments. On the assumption that these environments will support different sets of species, these classifications have then been used as broad scale biodiversity surrogates (e.g. Nix et al., 2000; Pressey et al., 2000; Thackway and Cresswell, 1995; Mackey et al., 1989). While such classifications can be efficient tools for planning and biodiversity assessment at relatively large scales (Pressey and Nicholls, 1989), they are often inaccurate at the scale of a patch or stand of vegetation. Consequently they are of limited value for guiding management decisions at this finer scale, such as identifying the need for weed control, monitoring grazing regimes, assessing the impact of firewood removal on coarse woody debris levels, or identifying the need for enrichment plantings to regenerate absent lifeforms. The lack of precision at this finer scale is because within any broad

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1 A stand is an area of forest or woodland whose structure or composition is different from adjacent areas (Lindenmayer and Franklin, 2002). A stand is comparable in scale to a patch.
environmental classification, such as a bioregion, environmental domain, or vegetation community, biodiversity can be highly variable as a result of past natural disturbance or modification by humans (e.g. Prober and Thiele, 1995). Stand scale indicators are critical for explaining this additional variation.

At the scale of a stand, biodiversity indicators are usually placed into one of two categories: those based on the identification of key species, and those based on the identification of key structures (Lindenmayer et al., 2000). This is not an absolute categorisation because there are approaches that utilise a combination of both key species and key structures as indicators of biodiversity. For example, Watson et al. (2001) demonstrated that particular bird species with demanding habitat requirements (focal species) could be used to identify the level of habitat structural complexity required to sustain all other woodland bird species. However, used in isolation from structural variables, key species have had limited success because robust relationships between potential indicator species, or groups of species, and total biodiversity have not been well established (Duelli and Obrist, 2003; Van Den Meersschaut and Vandekerkhove, 1998; Lindenmayer and Cunningham, 1997; Yen, 1987; Majer, 1983). On this basis, Margules et al. (2002) concluded there was no compelling evidence that groups of species could be used to represent biodiversity as a whole.

On the other hand, indicators based on key structures are generating considerable interest, both in their role as practical surrogates for biodiversity (Uuttera et al., 2000; Lahde et al., 1999; Koop et al., 1994; Buongiorno et al., 1994) and as a key to understanding the sources of biodiversity in forested ecosystems (Spies, 1998; Franklin, 1988; Franklin et al., 1981). The rationale for this approach is that ecosystems containing stands with a variety of structural components are considered likely to have a variety of resources and species that utilise these resources (Tanabe et al., 2001; Brokaw and Lent, 1999; Pretzsch, 1997; Woinarski et al., 1997). Consequently there is often a positive correlation between elements of biodiversity and measures of the variety and / or complexity of structural components within an ecosystem (e.g. MacNally et al., 2001).
Structural complexity necessarily involves the interaction between a number of different attributes (variables) so that quantitative comparisons between stands can require complex multivariate analysis (e.g. Spies and Franklin, 1991). In response to this problem, a variety of indices have been devised with the aim of expressing structural complexity as a single number, thereby facilitating comparisons between stands (e.g. Watson et al., 2001; Koop et al., 1994; Newsome and Catling, 1979). By acting as a summary variable for a larger pool of structural attributes, these indices can also provide a means of ranking stands in terms of their potential contribution to biodiversity (e.g. Parkes et al., 2003; Van Den Meersschaut and Vandekerkhove, 1998). An index of this type also facilitates the use of alternative policy instruments for biodiversity conservation such as mitigation banking, auctions and offsets, that rely on a common currency – the index value – that can be compared or traded between sites (e.g. Parkes et al., 2003; Chaudhri, 2003).

Stand based assessments of structural components are also important because they can inform on-ground management. This is because the stand is the scale at which management actions are usually implemented (O'Hara, 1998), and also because the effect of management on structural components can be direct and easy to assess (e.g. the removal of large hollow bearing trees) compared to assessing the impact on faunal species. In south-eastern Australia substantial amounts of money and regulatory control are currently being translated into on-ground actions via stand level assessments. These include the assessment of site condition as part of a market based mechanism for purchasing land management services by the Victorian Department of Sustainability and Environment (e.g. Parkes et al. 2003), and the use of site based assessments to determine whether an application for broadscale clearing of native vegetation improves or maintains environmental outcomes for biodiversity under the NSW Native Vegetation Act (2003) (e.g. Gibbons et al. 2004).
1.2 Research questions

Stand level assessments depend on the use of an appropriate metric to quantify the variety of structural components present. A variety of different metrics or structural complexity indices have been proposed for this purpose, reflecting the presence of a range of rationales and methodologies. The aim of this thesis is to bring some objectivity to this situation by establishing a clear and documentable methodology for developing a stand scale index of structural complexity, and then testing this methodology using data from real stands. To do this I intend to address the following research questions:

1. How should stand structure and stand structural complexity be defined?
2. Is there any consensus in the ecological literature as to how stand structural complexity should be quantified?
3. What key steps constitute a methodology for developing an index of structural complexity, so that it has a documentable relationship with stand scale elements of biodiversity?
4. Can this methodology be successfully applied to remnant native vegetation within an agricultural landscape?
5. Can the index developed using this methodology function as a practical tool for land managers?
6. How does the performance of this index compare to other prominent indices?

In the next chapter I address the first two of these questions by exploring in detail the nexus between biodiversity and stand structure. To do this I review the ecological literature, to provide a clear definition of forest and woodland stand structure, and to identify the range of stand level structural attributes used to characterise stand structure.
Chapter 2: Stand structure and structural attributes

Chapter summary
This chapter reviews the literature concerning forest and woodland structure at the scale of an individual stand. Stand structure is defined in terms of structural attributes and stand structural complexity. Stand structural complexity is considered to be a combined measure of the number of different attributes present and the relative abundance of each of these attributes. The review indicates there is no definitive suite of structural attributes; different authors emphasise subsets of different attributes, and relatively few studies provide quantitative evidence linking attributes to the provision of faunal habitat or other measures of biodiversity, although a number of studies identify attributes that distinguish between successional stages. A summary of key structural attributes identified in the literature is presented under the following stand elements: foliage arrangement, canopy cover, tree diameter, tree height, tree spacing, tree species, stand biomass, understorey vegetation, and deadwood.

2.1 Defining forest and woodland structure

2.1.1 Ecosystem structure, function and composition
Ecosystems are frequently characterised in terms of their species and genetic composition (Hunter, 1999; Claridge et al., 1997; Magurran, 1988), even though this approach ignores ecological processes (e.g. natural disturbance, the decomposition of woody debris, the cycling of nutrients etc.), which are critical for the maintenance of biodiversity (Noss, 1990). An alternative approach, first suggested more than two decades ago, (e.g. Franklin et al., 1981), is to describe forest ecosystems by attributes relating to ecosystem structure and function in addition to those describing composition (Franklin et al., 2002; Noss, 1990; Franklin, 1988). In this approach:

- Structure refers to the spatial arrangement of the various components of the ecosystem, such as the heights of different canopy levels and the spacing of trees;
- Function refers to how various ecological processes, such as the production of organic matter, are accomplished and to the rates at which they occur;
- Composition refers to the identity and variety of ecosystem components, as characterised by species richness and abundance.
2.1.2 Structural attributes

The structural, functional and compositional attributes of a stand are often interdependent, so that attributes from one group may also be surrogates for attributes from another group (Franklin et al., 2002; Ferris and Humphrey, 1999; Noss, 1990). For example a structural attribute such as dead wood can also be a good indicator of functional attributes such as decomposition and nutrient cycling processes (Franklin et al., 1981). Similarly, compositional attributes, such as species composition and abundance can be indicators of structural attributes such as canopy layering (Franklin et al., 2002), or of functional attributes such as flowering and bark shedding (Kavanagh, 1987). The division of attributes into three groupings is therefore by no means a clear categorisation. In order to define structure in an unambiguous manner, this paper will therefore pool structural, functional and compositional attributes into a single category simply called structural attributes.

2.1.3 Stand structure

Stand structure is commonly defined in terms of two components in the ecological literature - stand structural attributes and stand structural complexity. Stand structural attributes can include measures of:

- **Abundance** - e.g. density of large trees (Acker et al., 1998), volume of dead wood (Wickstrom and Erickson, 2000);
- **Relative abundance** - e.g. foliage height diversity (MacArthur and MacArthur, 1961), dbh \(^2\) diversity (Gove, 1996; Buongiorno et al., 1994), basal area of deciduous tree species (Spies and Franklin, 1991);
- **Richness** - e.g. overstorey species richness (Munks et al., 1996), eucalypt species richness (Bauer et al., 2000), shrub species richness (Seddon et al., 2001);
- **Size variation** - e.g. standard deviation of dbh (Spies and Franklin, 1991);
- **Spatial variation** - e.g. coefficient of variation of distance to nearest neighbour (Franklin et al., 1981);

Attributes that quantify variation are particularly important because these can

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\(^2\) dbh or diameter at breast height, is the diameter of a tree measured at a standard height above the ground on the uphill side of the tree. In Australia this standard height is 1.3m.
also describe habitat heterogeneity at the stand scale. For example the coefficient of variation of shrub cover would be a measure of the patchiness of understorey vegetation, a feature which is important for some macropod species (Lunney and Ashby, 1987).

Stand structural complexity is essentially a measure of the number of different structural attributes present and the relative abundance of each of these attributes. *Structural complexity* is used in preference to *structural diversity* because the latter term is considered ambiguous. This reflects the work of a number of authors in which a diversity measure, such as the Shannon-Weiner Index, has been used to quantify a single attribute, such as variation in stem diameter (e.g. Gove *et al.*, 1995; Buongiorno *et al.*, 1994). This quantity is then deemed a measure of structural diversity and to be indicative of biological diversity. In reality all that has been quantified is one of many possible attributes, and by most measures of diversity a system with one attribute or element has a diversity of zero (Magurran, 1988).

### 2.2 Stand structural attributes

#### 2.2.1 Overview of attributes

To be an efficient and effective biodiversity surrogate, any measure of structural complexity will need to be based on an appropriate suite of structural attributes. This suite should be sufficiently comprehensive to capture the variety of structural components that occur in forests and woodlands, reflect observed relationships with faunal diversity, and remain concise enough to function as a practical tool for land managers. In this section, I review the range of structural attributes used to characterise stand structure.

The studies that were reviewed and their associated attributes are summarised in Table 1. For clarity, attributes have been grouped in Table 1 under the stand element they aim to describe (e.g. foliage, tree diameter etc.). For studies where a large number of attributes were tested, only those attributes that proved significant in the modelling process have been included in Table 1.
Table 1: Attributes used to characterise stand structure. Attributes are grouped under the stand element they aim to describe. Superscript indicates an alternative measure used by a particular author.

<table>
<thead>
<tr>
<th>Stand element</th>
<th>Attribute</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Foliage</td>
<td>Foliage Height Diversity</td>
<td>Sullivan et al., 2001; Tanabe et al., 2001; Berger and Puettmann, 2000; Gove, 1996; MacArthur and MacArthur, 1961;</td>
</tr>
<tr>
<td></td>
<td>Number of strata</td>
<td>Tanabe et al., 2001; Van Den Meersschaut et al., 1998; Maltamo et al., 1997; Uuttera et al, 1997; Koop et al., 1994; MacArthur and MacArthur, 1961;</td>
</tr>
<tr>
<td></td>
<td>Foliage density within different strata</td>
<td>Bebi et al., 2001; Ferris-Kaan et al., 1996;</td>
</tr>
<tr>
<td>Canopy cover</td>
<td>Canopy cover</td>
<td>Parkes et al., 2003; Watson et al., 2001; Van Den Meersschaut et al., 1998; Newsome and Catling, 1979</td>
</tr>
<tr>
<td></td>
<td>Gap size classes</td>
<td>Tyrell and Crow, 1994;</td>
</tr>
<tr>
<td></td>
<td>Average gap size and the proportion of canopy in gaps</td>
<td>Tanabe et al., 2001; Ziegler, 2000; Tyrell and Crow, 1994;</td>
</tr>
<tr>
<td></td>
<td>Proportion of tree crowns with broken and dead tops</td>
<td>Spies and Franklin, 1991;</td>
</tr>
<tr>
<td>Tree diameter</td>
<td>Tree dbh</td>
<td>Tanabe et al., 2001; Ziegler, 2000; Ferreira et al., 1999; Uuttera et al., 1997; Acker et al., 1998; Spies and Franklin, 1991;</td>
</tr>
<tr>
<td></td>
<td>Standard deviation of dbh (coefficient of variation(^A))</td>
<td>Zenner, 2000; Acker et al., 1998; Van Den Meersschaut et al., 1998; Spies and Franklin, 1991; Franklin et al., 1981;</td>
</tr>
<tr>
<td></td>
<td>Tree size diversity</td>
<td>Wikstrom and Erickson, 2000; Gove, 1996; Buongiorno et al., 1994;</td>
</tr>
<tr>
<td></td>
<td>Horizontal variation in dbh</td>
<td>Zenner, 2000;</td>
</tr>
<tr>
<td></td>
<td>Diameter distribution</td>
<td>Bachofen and Zingg, 2001; Uuttera et al., 2000; Ferreira and Prance, 1999; Maltamo et al., 1997; Kappelle et al., 1996; Tyrell and Crow, 1994; Koop et al., 1994;</td>
</tr>
<tr>
<td></td>
<td>Number of large trees</td>
<td>Ziegler, 2000; Acker et al., 1998; Van Den Meersschaut et al., 1998; Tyrell and Crow, 1994; Koop et al., 1994; Spies and Franklin, 1991;</td>
</tr>
<tr>
<td>Tree height</td>
<td>Height of overstorey</td>
<td>Dewalt et al., 2003; Bebi et al., 2001; Means et al., 1999; Spies, 1998; Kappelle et al., 1996; Koop et al., 1994;</td>
</tr>
<tr>
<td></td>
<td>Standard deviation of tree height</td>
<td>Zenner, 2000;</td>
</tr>
<tr>
<td></td>
<td>Horizontal variation in height</td>
<td>Svensson and Jeglum, 2001; Zenner, 2000;</td>
</tr>
<tr>
<td></td>
<td>Height class richness</td>
<td>Sullivan et al., 2001;</td>
</tr>
<tr>
<td>Tree spacing</td>
<td>Clark Evans Index, Cox Index(^B), percentage of trees in clusters(^C)</td>
<td>Bachofen and Zingg, 2001; Svensson and Jeglum, 2001; Bebi et al., 2001; Zenner, 2000; Pretzsch, 1997;</td>
</tr>
<tr>
<td></td>
<td>Number of trees per ha</td>
<td>Bachofen and Zingg, 2001; Uuttera et al., 2000; Ferreira and Prance, 1999; Acker et al., 1998; Kappelle et al., 1996; Spies and Franklin, 1991;</td>
</tr>
</tbody>
</table>
(Table 1 continued)

<table>
<thead>
<tr>
<th>Stand element</th>
<th>Attribute</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Stand biomass</strong></td>
<td>Stand basal area</td>
<td>Denslow and Guzman, 2000; Ziegler, 2000; Means et al., 1999; Ferreira and Prance, 1999; Kappelle et al., 1996; Koop et al., 1994; Means et al., 1999; Ferreira and Prance, 1999; Spies, 1998;</td>
</tr>
<tr>
<td></td>
<td>Stand volume</td>
<td>Uuttera et al., 2000; Means et al., 1999; Ferreira and Prance, 1999; Spies, 1998;</td>
</tr>
<tr>
<td><strong>Tree species</strong></td>
<td>Species diversity and / or richness</td>
<td>Sullivan et al., 2001; Uuttera et al., 2000; Van Den Meersschaut et al., 1998; Maltamo et al., 1997; Uuttera et al., 1997;</td>
</tr>
<tr>
<td></td>
<td>Relative abundance of key species</td>
<td>Berger and Puettmann, 2000; Ziegler, 2000; Wikstrom and Erickson, 2000; Spies and Franklin, 1991;</td>
</tr>
<tr>
<td><strong>Understorey vegetation</strong></td>
<td>Herbaceous cover and/or its variation</td>
<td>Parkes et al., 2003; Watson et al., 2001; Van Den Meersschaut et al., 1998; Spies and Franklin, 1991; Newsome and Catling, 1979;</td>
</tr>
<tr>
<td></td>
<td>Shrub cover</td>
<td>Parkes et al., 2003; Watson et al., 2001; Berger and Puettmann, 2000; Spies and Franklin, 1991; Newsome and Catling, 1979;</td>
</tr>
<tr>
<td></td>
<td>Shrub height</td>
<td>Berger and Puettmann, 2000;</td>
</tr>
<tr>
<td></td>
<td>Total cover of understorey</td>
<td>Spies and Franklin, 1991;</td>
</tr>
<tr>
<td></td>
<td>Understorey richness</td>
<td>Sullivan et al., 2001; Van Den Meersschaut et al., 1998;</td>
</tr>
<tr>
<td></td>
<td>Understorey stems (Sapling) per ha</td>
<td>Dewalt et al., 2003; Van Den Meersschaut et al., 1998; Spies and Franklin, 1991;</td>
</tr>
<tr>
<td><strong>Deadwood</strong></td>
<td>Number, volume or basal area of stags (by decay classes, by diameter class)</td>
<td>Dewalt et al., 2003; Bachofen and Zingg, 2001; Svennson and Jeglum, 2001; Sullivan et al., 2001; Van Den Meersschaut et al., 1998; Tyrell and Crow, 1994; Spies and Franklin, 1991; Franklin et al., 1981;</td>
</tr>
<tr>
<td></td>
<td>Volume of coarse woody debris</td>
<td>Sullivan et al., 2001; Svennson and Jeglum, 2001; Ziegler et al., 2000; Wikstrom and Erickson, 2000; Tyrell and Crow, 1994;</td>
</tr>
<tr>
<td></td>
<td>Log volume by decay or diameter classes</td>
<td>Dewalt et al., 2003; Van Den Meersschaut et al., 1998; Tyrell and Crow, 1994; Spies and Franklin, 1991; Franklin et al., 1981;</td>
</tr>
<tr>
<td></td>
<td>Log length or cover</td>
<td>Parkes et al., 2003; Watson et al., 2001; Newsome and Catling, 1979;</td>
</tr>
<tr>
<td></td>
<td>Coefficient of variation of log density</td>
<td>Spies and Franklin, 1991;</td>
</tr>
<tr>
<td></td>
<td>Litter biomass or cover</td>
<td>Denslow and Guzman, 2000; Parkes et al., 2003; Watson et al., 2001; Newsome and Catling, 1979;</td>
</tr>
</tbody>
</table>
The choice of attributes varied considerably in the studies reviewed. Different authors emphasised subsets of different attributes, and very few studies advocated similar suites of attributes. Only a few studies contained extensive sets of attributes (e.g. Van Den Meersschaert and Vandekerkhove, 1998; Spies and Franklin, 1991; Franklin et al., 1981), and no single study was considered likely to provide a definitive suite of attributes capable of characterising the full biodiversity potential of a stand. This lack of consensus may in part reflect differing rationales for selecting attributes in the first place. Some authors, for example, chose attributes because they were useful for purposes other than characterising biodiversity, such as predicting the protective function of forests in avalanche prone areas (e.g. Bebi et al., 2001; Bachofen and Zingg, 2001), or characterising the habitat of a particular faunal group (e.g. Watson et al., 2001; Newsome and Catling, 1979). Other authors chose attributes because they were relatively easy to measure and model, and were thought to be correlated with more complex attributes (e.g. Wikstrom and Eriksson, 2000; Gove, 1996; Buongiorno et al., 1994). In some cases a limited suite of attributes was utilised because of historical precedent, such as systematic measurements carried out in permanent plots (Acker et al., 1998; Maltamo et al., 1997), or because attributes reflected the current state of a particular remote sensing technology (e.g. Means et al., 1999).

In most cases attributes were assumed to be indicative of biodiversity, although relatively few studies provided quantitative evidence linking attributes to the provision of faunal habitat (Franklin et al., 1981 is a notable exception). A number of studies identify attributes that successfully distinguished between different successional stages (e.g. Dewalt et al., 2003; Ziegler, 2000; Tyrrell and Crow, 1994; Spies and Franklin, 1991). These may be important for biodiversity conservation since attributes that are changing or developing during succession are likely to be those responsible for the different habitats associated with each successional stage.

In the following sections, the relevance of each stand element and its associated attributes is discussed in detail. I also indicate why particular authors chose to use these attributes.
2.2.2 Foliage

The vertical arrangement of foliage is one of the attributes most commonly used to describe stand structure (Brokaw and Lent, 1999), and the first in which a quantitative relationship was established between an element of structure and a measure of faunal diversity. In a landmark study, MacArthur and MacArthur (1961) established a linear relationship between Foliage Height Diversity (FHD), which described the arrangement of foliage within different vertical strata, and Bird Species Diversity (BSD). FHD was based on an index developed by communications engineers (Hurlbert, 1971), and was defined by the relationship:

\[ \text{FHD} = -\sum p_i \ln p_i \]

where \( p_i \) is the proportion of total foliage which lies in the ith of the chosen foliage layers. BSD was described using a similar approach where

\[ \text{BSD} = -\sum p_i \ln p_i \]

and \( p_i \) is the proportion of all bird individuals which belong to the ith species, and \( \ln \) denotes natural logarithm.

By a process of trial and error MacArthur and MacArthur (1961) found that the three height classes 0 - 0.61m, 0.61m – 7.62m and >7.62m, gave the best correlation with BSD when used in their FHD index. These divisions also corresponded to the layers occupied by the foliage of herbs, shrubs and trees respectively. The authors concluded that it was the distribution of foliage amongst different life forms, which provided more kinds of habitat, and thereby supported a greater diversity of birds.

MacArthur and MacArthur’s results have been supported by a number of subsequent studies (Ambuel and Temple, 1983; Abbott, 1976; Recher, 1969). This has encouraged the use of FHD as a measure of forest structure and its acceptance by some authors as a reliable indicator of biodiversity (Tanabe et al., 2001; Gove, 1996; Buongiorno et al., 1994), although there is little evidence to suggest that FHD can explain differences in the diversity of faunal groups other than birds.

Despite its acceptance by some authors, FHD appears as an ambiguous measure in the literature because no standard method has been established for its measurement. When calculating FHD, different studies have assessed...
vegetation density in different ways (Parker and Brown, 2000; Erdelen, 1984), and unlike MacArthur and MacArthur’s original study in which class intervals related to distinct life forms, the delineation of height classes is often arbitrary (e.g. Sullivan et al., 2001; Tanabe et al., 2001). The precision of these various approaches and their correlation with each other is unclear and limits comparisons between studies (Erdelen, 1984).

Alternatives to FHD as measures of foliage structure include:

- Comparing foliage cover for selected species in selected strata (Bebi et al., 2001) or comparing the relative cover for selected species in two strata - for example, the ratio of lower canopy cover to ground foliage cover (Ferris-Kaan et al., 1996);
- Specifying the number of strata on the basis that multi-layered stands increase the diversity of habitat niches (Van Den Meersschaut and Vandekerkhove, 1998);
- Describing the amount of foliage cover in height classes relating to distinct strata (Watson et al., 2001; Catling and Newsome, 1979).

Each of these approaches assumes that the strata within a canopy can be clearly and consistently defined. Parker and Brown (2000) reviewed the concept of "canopy stratification" in the ecological literature, and demonstrated that the term had a variety of meanings that were both difficult to measure and to reconcile. As a practical demonstration of this ambiguity, they applied a number of definitions of stratification to a mixed-oak forest canopy in south-western Virginia, USA. These definitions failed to consistently and clearly identify the presence, number or location of strata. Parker and Brown (2000) attributed this to the presence of trees of different shade tolerance, which formed a continuous canopy from ground layer to canopy top. They suggested that it would be more meaningful to define strata in terms of the ecological gradients created by foliage (e.g. changes in light transmittance levels) rather than attempt to describe the arrangement of the foliage itself.

However, for ecosystems containing a single overstorey layer of shade intolerant species, such as the dry sclerophyll forests and woodlands of south-
eastern Australia, distinct strata are often clearly defined because foliage layers
tend to reflect different life forms (trees, shrubs, grasses, forbs) rather than
different shade tolerances within the same life form (Florence, 1996). For these
ecosystems a number of authors have described foliage arrangement by the
amount of cover in height classes corresponding to the different life forms that
are present (Watson et al., 2001; Catling and Newsome, 1979); an approach
which is consistent with the conclusions reached by MacArthur and MacArthur
(1961).

In summary it appears that although the use of vertical foliage arrangement to
classify stand structure is common in the literature, there is little consensus
as to what measure to use for this purpose. A related, and more straightforward
approach has been to characterise structure in terms of canopy cover, and the
variation in that cover that is produced by canopy gaps.

2.2.3 Canopy cover

Canopy cover is commonly defined as the percentage cover of the vertical
projection of tree crowns, where the crowns are considered as opaque objects
(Walker and Hopkins, 1990). While it is simpler to calculate than projective
foliage cover, canopy cover is also less meaningful as a biological measure.
This is because overstorey species differ in the opaqueness of their crowns,
and therefore in the amount of light they either intercept for their own primary
production, or else transmit for use by understorey vegetation.

Distinct changes have been observed in canopy cover during stand
development. For example in even-aged stands canopy cover will generally
increase from a low level at stand initiation, reaching a maximum shortly after
canopy closure at the beginning of the stem exclusion stage, then declining
again as overstorey elements disintegrate and canopy gaps form during the old-
growth stage (Franklin et al., 2002; Oliver and Larsen, 1996). Van Den
Meersschaut and Vandekerkhove (1998) acknowledged this dynamic when
quantifying canopy cover as a component of an index of forest structure for
closed forests in Belgium. They gave the maximum score in their index to
stands with 1/3 to 2/3 of potential maximum canopy cover.
Canopy gaps are an important structural attribute associated with variance in canopy cover. In North America, Ziegler (2000) and Tyrrell and Crow (1994), found that the proportion of the canopy in gaps, the average size of gaps and the distribution of gaps amongst size classes were all important attributes for distinguishing old-growth hemlock (*Tsuga canadensis*) hardwood forests from earlier successional stages. Similarly, Spies and Franklin (1991) found that the number of trees with dead tops or broken crowns was a key attribute that distinguished between old-growth, mature and young stands in Douglas-fir (*Pseudotsuga menziesii*) forests.

While the extent and condition of canopy cover are important attributes of forest structure, they appear relatively infrequently in the literature. The focus for many studies has been the use of attributes related to the size of tree stems such as tree diameter, height and volume. This in part reflects the association between stem size and other structural attributes such as foliage distribution, crown dimensions and the production of dead wood of different sizes (Spies, 1998). It also reflects the fact that data on stem size are relatively straightforward to collect and have historically formed the basis of many permanent plot measurements (e.g. Acker et al., 1998).

### 2.2.4 Tree diameter

Tree diameter is a ubiquitous measure of tree size in the literature concerning forest structure. It is typically quantified in terms of mean dbh, the standard deviation of dbh, the number of trees exceeding a threshold diameter (i.e. the number of large trees) and the frequency distribution of dbh classes. Spies and Franklin (1991) identified the first three of these structural attributes as amongst the most important in a range of attributes they tested for characterising wildlife habitat, ecosystem function and successional development in Douglas-fir forests.

*Mean dbh*

Mean dbh generally increases with stand age and has been used to discriminate between successional stages in Douglas-fir forests (Spies and Franklin, 1991), hemlock-hardwood forests (Ziegler, 2000) and between managed and virgin boreal forests in Finnish and Russian Karelia (Uuttera et
al., 1997). However, Franklin et al. (1981) found that old-growth and young stands of Douglas-fir had a similar mean dbh, although the old-growth stand had nearly twice the coefficient of variation of dbh compared with the young stand.

Stand basal area is an attribute directly related to mean dbh. It is also indicative of stand volume and biomass. Stand basal area has been used to discriminate between primary and secondary Quercus forest in Costa Rica (Kappelle et al., 1996), and successional stages in hemlock-hardwood forests (Ziegler, 2000).

The standard deviation of tree dbh

The standard deviation of tree dbh is a measure of the variability in tree size, and is considered indicative of the diversity of micro-habitats within a stand (Acker et al., 1998; Van Den Meerschaut and Vandekerkhove, 1998). It is a straightforward attribute to quantify and can be comparable with more complex attributes and indices as a descriptor of stand structure (Neumann and Starlinger, 2001). For example Spies and Franklin (1991) found that the standard deviation of dbh was more useful than a measure of height diversity in discriminating between successional stages of Douglas-fir forests. Similarly, Zenner (2000) found that a Structural Complexity Index based on a three dimensional model of forest structure was significantly correlated with the standard deviation of dbh, and Neumann and Starlinger (2001) found that the standard deviation of dbh was significantly correlated with seven indices of structural complexity they tested.

Number of large trees

The number of large trees is indicative of the potential number of occupied hollow bearing trees, dead standing trees, and logs. For example, in a review of 13 different studies, which addressed the occupancy of trees by hollow dependent fauna, Gibbons et al. (2002), found that 80% of studies reported a significant correlation between occupancy and tree dbh. While trees with larger dbh are likely to contain more occupied hollows, the majority of hollow-bearing trees in a stand usually occur in the middle diameter classes, rather than in the largest trees (Gibbons and Lindenmayer, 2002).
Large trees have also been associated with four key structural attributes distinguishing old-growth Douglas-fir forests from earlier successional stages. Franklin et al. (1981) identified these as:

1. Large living old-growth trees with dbh >100 cm;
2. Large dead standing trees;
3. Large logs on land and;
4. Large logs in streams.

The number of large trees has been used to characterise the structure of hemlock-hardwood stands (Ziegler, 2000; Tyrrell and Crow, 1994), Douglas-fir stands (Acker et al., 1998; Spies and Franklin, 1991; Franklin et al., 1981), temperate forests in Belgium (Van Den Meerschaut and Vandekerkhove, 1998) and Australia (Parkes et al., 2003), Sumatran rainforest (Koop et al., 1994) and moist tropical forest in central Panama (Dewalt et al., 2003). The threshold diameter used to define a large tree varies between these studies, ranging from > 65 cm in the case of Dewalt et al. (2003), to > 100 cm dbh in the case of Spies and Franklin (1991). The ecological basis for these thresholds is unclear, although they presumably reflect potential growth over a time frame required for individual trees to become mature and to develop structural characteristics such as stem hollows and fragmented crowns. The inherent variation between species in the dbh at which this occurs necessitates the use of different thresholds for different site and forest types.

**Frequency distribution of dbh classes**

Quantifying the number of large trees essentially places trees into two diameter classes. A number of different authors have taken the next logical step of using a frequency distribution across a range of diameter classes as an attribute of stand structure. Koop et al. (1994) applied this approach in a simple form. Stands with a reverse J distribution, indicative of uneven-aged stands, scored 2 points on an index of structure, whereas any other distribution scored 1 point. A more elaborate approach was adopted by Tyrell and Crow (1994), who used tree numbers across 10 cm diameter classes to characterise the structure of hemlock-hardwood stands. They found that old-growth stands were distinguished by a relatively even diameter distribution, with low densities of
trees across all diameter classes. Dewalt et al. (2003) used a similar approach to characterise the diameter distribution of stands from a chronosequence in Panamanian moist tropical forest. They found it took secondary stands 70-100 years to develop a diameter distribution similar to old-growth stands.

This contrasts with a number of other studies in tropical forests in which diameter distribution did not appear as a particularly useful or significant attribute with which to characterise forest structure (Uuttera et al., 2000; Ferreira and Prance, 1999; Kappelle et al., 1996). In these forests, management often removes large trees of commercial species, so that changes in structure may be better characterised by assessing a few species in particular diameter classes rather than the entire diameter distribution (Uuttera et al., 2000).

One of the drawbacks to using diameter distribution as an attribute of forest structure is the complexity of comparing distributions from different stands. For example, Faith et al. (1985) developed a classification algorithm to compare the diameter distributions of different stands of *Eucalyptus sieberi*. While their algorithm successfully identified stands with similar logging and disturbance histories, it did so by summarising the differences between diameter distributions rather than the distribution per se. A number of authors have addressed this issue by using the Shannon-Weiner Index to summarise diameter distribution in a single measure called tree size diversity (H'); where \( H' = -\sum (p_i \ln p_i) \); and \( p_i \) is the proportion of trees in the \( i \)th diameter class (Wikstrom and Erickson, 2000; Gove, 1996; Buongiorno et al., 1994). This measure permits direct comparisons between diameter distributions, and also provides a simple attribute that can be modelled under different management strategies. For example, Buongiorno et al. (1994) modelled the relationship between tree size diversity and sustainable timber revenue in northern hardwood stands in Wisconsin, USA.

While mathematically expedient, the ecological rationale for using tree size diversity as an attribute of structure is not clearly established in the literature. Intuitively it appears to be a functional attribute for quantifying the presence of
continuous regeneration and healthy stand replacement, although no author acknowledges this explicitly. Buongiorno et al. (1994) claim tree size diversity to be a good proxy for foliage height diversity, on the basis that a diversity of tree sizes will equate with a diversity of canopy heights and foliage arrangements. However, this argument is flawed because foliage height diversity is sensitive to the way in which height classes are defined (Parker and Brown 2000). MacArthur and MacArthur (1961) used unequal height classes that equated to different life forms. Tree size diversity on the other hand establishes arbitrary classes based on equal diameter class widths rather than observable strata.

In contrast to Buongiorno et al. (1994), Gove (1996) drew no connection between foliage height diversity and tree size diversity. Instead, Gove (1996) used these measures to characterise different attributes of the structure of Spruce-Fir stands in Maine. Foliage height diversity was used to characterise vertical structure and tree size diversity was used to characterise horizontal structure. However, as Zenner (2000) demonstrates, stands with the same diameter distribution - and hence tree size diversity - can have trees in very different spatial arrangements. It would therefore appear that the use of tree size diversity as a structural attribute has more to do with it being a quantity which is easy to compute and model rather than with established connections to stand structure or biodiversity.

### 2.2.5 Tree height

Quantitative relationships between tree height and diameter are well established in the literature (Temesgen and Gadow, 2004), so that to some extent structural attributes associated with diameter may also serve as proxies for attributes associated with tree height (e.g. Buongiorno et al., 1994). However, because the relationship between height and diameter is non-linear, it is often more meaningful to use attributes directly associated with height when characterising vertical elements of structure. For example the standard deviation of tree height will be more indicative of the vertical layering of foliage than the standard deviation of dbh (Zenner, 2000).

The simplest attribute associated with height is the height of the overstorey. It is
an attribute, which is readily derived from remotely sensed data (e.g. Bebi et al., 2001; Means et al., 1999), and can be indicative of successional stage (Kappelle et al., 1996), the number of strata (Spies, 1998) and stand biomass (Means et al., 1999).

Variation in tree height is considered an important attribute of structure because stands containing a variety of tree heights are also likely to contain a variety of tree ages and species thereby providing a diversity of micro-habitats for wildlife (Zenner, 2000). Sullivan et al. (2001) quantified this type of variation in terms of a simple measure called structural richness, which was based on the number of height classes occupied by the trees in the stand.

Zenner (2000) and Svensson and Jeglum (2001) indicated that variation in tree height is more complex than structural richness, because it depends on the horizontal arrangement of the trees as well as the height of the trees. Zenner (2000) therefore used a three-dimensional model of the position of trees to describe variation in tree height in terms of a structural complexity index. This index is discussed in Chapter 3.

2.2.6 Tree spacing

In contrast to the approach of Zenner (2000), a number of authors have used the horizontal distribution of trees as a structural attribute in its own right. The simplest way to do this is to use the number of trees per ha as a measure of the average space between trees. This attribute has successfully distinguished between successional stages of unmanaged Douglas-fir forests (Spies and Franklin, 1991), and primary and secondary tropical forest in central Amazonia (Ferreira and Prance, 1999). However, a number of other studies in tropical forest systems have not confirmed that number of trees per ha is a useful attribute, and in these forests it failed to distinguish between the structure of primary and secondary forest (Uuttera et al., 2000; Kappelle et al., 1996).

A more complex approach is to describe tree spacing in terms of its variation rather than its average dimension. This is likely to be more meaningful because it is indicative of the size and distribution of gaps, and these are critical to
processes such as recruitment, competition and mortality (Svensson and Jeglum, 2001). Typically variation in spacing is described by an aggregation index, which quantifies the degree of clustering in the horizontal arrangement of trees; Pommerening (2002) provides a useful review of such indices.

The most commonly used aggregation index is the Clark-Evans index (Pretzsch, 1997). This index describes the ratio between the observed average distance of a tree to its nearest neighbour and the expected average distance for a random tree distribution. Aggregation values below 1 are indicative of a tendency towards clustering, values close to 1 indicate a random distribution and values greater than 1 are indicative of an increasingly regular distribution (Clark and Evans, 1954). Bachofen and Zingg (2001) found that the Clark-Evans index was one of the three most useful attributes for detecting structural changes in subalpine Norway Spruce (Picea abies) forests. A major problem with applying this index is that it requires nearest neighbour distances to be determined for each tree in the plot.

The Cox Index of clumping addresses this issue by quantifying the variance to mean ratio of the number of trees in equal sized sub-plots rather than using nearest neighbour distances (Neumann and Starlinger, 2001). A variance to mean ratio of 1 indicates a random distribution, values less than 1 a regular distribution and values greater than 1 an increasingly clustered distribution. Svensson and Jeglum (2001) applied the variance to mean ratio approach to an undisturbed old-growth Norway Spruce forest. They found a clear difference between the spatial distribution of the overstorey and understorey trees. Trees in the overstorey were randomly distributed, whereas trees in the understorey were clumped. The clumping reflected the location of understorey trees in the spaces between overstorey trees. The authors concluded that a key characteristic of old-growth Norway Spruce forest was a disorderly three dimensional structure and that this was well described by the variance to mean ratio of tree numbers (Svensson and Jeglum, 2001). Franklin et al. (1981) described a similar effect in old-growth Douglas-fir forests in which groups of small diameter shade tolerant trees developed under an overstorey of large well spaced Douglas-firs. Lindenmayer et al. (1990a) used the variance to mean
ratio to quantify the pattern of hollow-bearing trees in Mountain Ash (*Eucalyptus regnans*) forests in south-eastern Australia. They found that the occupancy of hollows was limited when trees were clustered rather than more uniformly spaced.

### 2.2.7 Tree species

Many of the attributes described in the preceding sections can be related in some degree to the presence of a variety of tree species. For example the presence of a mix of shade-tolerant and shade-intolerant tree species is likely to produce a multi-layered canopy, a large variety of stem diameters and heights, and a complex horizontal arrangement of stems (e.g. Spies and Franklin, 1991). It is therefore not surprising that the literature contains a number of studies in which measures of tree species richness and/or diversity have been used to characterise stand structure (e.g. Sullivan *et al.*, 2001; Maltamo, 1997; Uuttera *et al.*, 1997).

Species richness and diversity are particularly important attributes of tropical forests, and in some cases are more successful in distinguishing between primary and secondary forest than attributes such as mean diameter or number of trees (Uuttera *et al.*, 2000; Ferreira and Prance, 1999), although Dewalt *et al.* (2003) found no differences in the overstorey richness of Panamanian primary and secondary tropical forests. This contrasts with some temperate forests where it is the relative abundance of key tree species or groups of species, rather than species richness, which distinguishes successional stages. For example Ziegler (2000) found that old-growth hemlock-hardwood stands had a higher proportion of hemlock trees than regrowth stands. Similarly Spies and Franklin (1991) found that the density of shade intolerant tree species was the most significant explanatory variable in a multiple regression relating stand age to a range of structural attributes.

### 2.2.8 Understorey vegetation

Up to this point this review has focused primarily on structural attributes associated with the trees in a stand, such as canopy layering, tree size variation, tree species richness, and the spatial arrangement of trees. This reflects a preoccupation in the literature with characterising structure in terms of
overstorey elements, with relatively little regard for the importance of other life forms. This can be partly justified on the grounds that overstorey structural attributes influence the condition of the understorey. For example Berger and Puettmann (2000) found that in aspen (*Populus tremuloides*) dominated forests herbaceous diversity was related to overstorey composition and to a measure of foliage height diversity, although shrub height and shrub cover were also important in explaining herbaceous diversity. A few notable studies have addressed the understorey directly and these are discussed below.

Spies and Franklin (1991) used 84 attributes to characterise the structure of unmanaged Douglas-fir forests. Fourteen of these attributes related directly to the understorey, and 6 of the 14 were significant in distinguishing between successional stages of Douglas-fir. In order of importance the understorey attributes were, % herbaceous layer cover, density of shade tolerant saplings, % cover of deciduous shrubs, density of subcanopy saplings, % total understorey cover, and coefficient of variation of herb cover.

Van Den Meersschaut and Vandekerkhove (1998) placed a similar emphasis on the understorey when developing a stand scale biodiversity index for forests in Belgium. In their index understorey attributes accounted for 25% of the total score, however unlike Spies and Franklin (1991) they placed relatively little importance on the amount of cover of the herb layer (3 out of 25). The highest weighting was given to the richness of understorey species (vascular species 10 out of 25, bryophytes 5 out of 25). Sullivan *et al.* (2001) also emphasised understorey species richness and diversity when characterising the structure of coniferous forests in the Pacific north-west. In their study herb and shrub species richness and diversity were treated as separate attributes of structure.

Understorey attributes are also prominent in indices used to characterise the structural complexity of temperate Australian forests and woodlands (Parkes *et al*., 2003; Watson *et al*., 2001; Newsome and Catling, 1979; Barnett *et al*., 1978). For example shrub and ground herbage cover account for 50% of the Habitat Complexity Score used by Watson *et al.* (2001), and the cover and richness of understorey life forms account for 25% of the Habitat Score.
proposed by Parkes et al. (2003). This emphasis is consistent with the shade-intolerant nature of the eucalypt species that dominate the overstorey of many Australian ecosystems, and whose relatively open crowns maintain a well lit understory (Bauhus et al., 2001; Florence, 1996). It also reflects the high proportion of plant species richness that is held in the understory of these ecosystems, and the relative sensitivity of the understory to modification.

### 2.2.9 Dead wood

In contrast to understory attributes dead wood appears in the literature as a structural attribute comparable in importance with the overstorey trees. In their study of old-growth Douglas-fir forests Franklin et al. (1981: 41) concluded that, "To a large degree, success in managing forests for old-growth attributes will depend on learning to manage the dead organic matter (snags, logs and litter) as cleverly as the live trees." They attributed the distinctive features of old-growth Douglas-fir forests to four structural elements, three of which related to dead wood in the form of:

- large dead standing trees;
- large logs on land and;
- large logs in streams.

**Standing dead trees**

The importance of standing dead trees as a structural attribute has been confirmed in subsequent studies in a variety of forest types (Dewalt et al., 2003; Bachofen and Zingg, 2001; Svennson and Jeglum, 2001; Sullivan et al., 2001; Van Den Meersschaut and Vandekerkhove, 1998; Tyrell and Crow, 1994; Spies and Franklin, 1991). In particular, standing dead trees can be an important source of tree hollows; for example Gibbons et al. (2000) found that 18% of hollow bearing trees in temperate forests in south-eastern Australia were dead trees, and Gibbons and Lindenmayer (2002) indicate that some hollow-nesting species have a preference for hollows in standing dead trees.

Despite their significance there is little consensus as to how to quantify standing dead trees. Franklin et al. (1981) quantified standing dead trees in old-growth Douglas-fir forests in terms of numbers per ha and mean dbh, and described
but did not apply a system suggested by Cline (1980) for classifying standing
dead trees into 5 decay classes. Franklin et al. (1981) considered decay
classes to be an important attribute because different vertebrates utilised trees
from different classes – a finding supported by Lindenmayer et al. (1990a) for
arboreal marsupials in the Mountain Ash forests of the central highlands of
Victoria.

In their study of unmanaged Douglas-fir forests Spies and Franklin (1991) used
a more elaborate set of attributes to describe standing dead trees. This included
number of standing dead trees per ha across two height classes and two
diameter classes, standing dead tree volume across 5 decay classes, total
standing dead tree volume, and the coefficient of variation of total standing
dead tree volume. However one attribute, the number of standing dead trees 5-
15m tall, explained most of the variation between successional stages that was
attributable to standing dead wood.

Tyrrell and Crow (1994) also quantified standing dead trees in detail in order to
characterise the structure of old-growth hemlock-hardwood forests. They
described standing dead trees in terms of species, dbh, four height classes,
three decay classes, basal area and total volume. The most useful of these was
the basal area of standing dead trees, which increased linearly with stand age.

Elaborate descriptions of standing dead tree condition may not therefore be
particularly useful. A more practical approach may be to simply quantify
standing dead trees in terms of their size and number. For example Van Den
Meersschaut and Vandekerkhove (1998) assessed standing dead trees in
terms of three simple attributes:
1. Basal area of standing dead trees - to indicate the volume of dead wood;
2. Number of large standing dead trees (dbh > 40cm) - to indicate the potential
   hollow bearing resource;
3. Standard deviation of standing dead tree dbh - to indicate the variation in size
   of dead wood and associated hollows, and hence its value to a diversity of
   fauna.
Logs

Standing dead trees are an important source of logs and so to some extent measures that quantify standing dead trees may also be indicative of the potential log resource. Svennson and Jeglum (2001) acknowledged this relationship by describing dead wood in terms of 5 types based on whether or not a log existed which could be connected to a standing dead tree or stump.

However, most studies that quantify dead wood treat logs separately to standing dead trees, because live trees are also an important source of logs, and logs may be removed from a stand by firewood collection or as a result of wildfire. The potential for a site to recruit logs is therefore unlikely to be a reliable measure of this structural component. A wide variety of attributes have been used to quantify logs including, total log length, total log volume, log volume or density by diameter or decay class, and the coefficients of variation in these attributes (e.g. Dewalt et al., 2003; Tyrrell and Crow, 1994; Spies and Franklin, 1991). The most useful of these appear to be those attributes describing the abundance of large logs, and variation in log numbers and size. For example, of the 10 attributes used by Spies and Franklin (1991) to describe logs, only three (>60cm diameter log density, >60cm log density coefficient of variation and total log volume) were significant in a canonical discriminant analysis used to relate age classes of unmanaged Douglas-fir to a suite of structural variables. Similarly Van Den Meersshaut and Vandekerkhove (1998) described logs in their index of biodiversity in terms of the total length of large logs (>40cm) and the number of diameter classes present. Parkes et al. (2003) also included a large log component in their Habitat Score. However rather than using a fixed threshold they specified logs in terms of the length of logs exceeding half the threshold diameter set for large trees. This threshold varied according to the vegetation community involved, on the basis that more productive ecosystems produce larger trees, and hence larger logs, than less productive ecosystems.
2.3 Conclusion
The review presented in this chapter demonstrates that there is a great variety of possible attributes for characterising stand structure. As a group, the attributes identified in the review describe stand structure in terms of foliage arrangement, canopy cover, tree diameter, tree height, tree spacing, tree species, stand biomass, understorey vegetation, and deadwood. To quantify structural complexity these many different attributes need to be considered simultaneously. This is because it is the combined effect of the number of different structural attributes present and the relative abundance of each of these attributes that endows a stand with its level of structural complexity. Quantifying structural complexity can therefore require complex multivariate analysis unless an overarching index is used to summarise the interaction of attributes in a single number. This is not a trivial task, requiring amongst other things deciding which of many possible structural attributes to use, and identifying an appropriate mathematical framework for synthesising these attributes in an index value. As a guide to how this might be done, in the next chapter, I review the various indices that have been used to characterise stand structural complexity.
Chapter 3: Indices of stand structural complexity

Chapter summary
This chapter reviews indices of stand structural complexity. The review identifies three types of index according to the mathematical framework underpinning the index. These are: additive indices based on the cumulative score of attributes; indices based on the average score of groups of attributes; and indices based on the interaction of attributes. The review identifies a variety of different indices under each of these frameworks with no single index preferred over the others. The most prominent of these indices are discussed in detail and the following guidelines suggested for the development of an index of structural complexity: 1. Start with a comprehensive set of attributes, in which there is a demonstrated association between attributes and the elements of biodiversity that are of interest. 2. Use a simple mathematical system to construct the index; this facilitates the use of multiple attributes and interpretation of the index in terms of real stand conditions. 3. Score attributes relative to the range of values occurring in stands of a comparable vegetation community. 4. Trial different weightings of attributes in the index, adopting those weightings which most clearly distinguish between stands. The chapter concludes by incorporating these guidelines into a methodology for developing an index of structural complexity.

3.1 Stand level indices of structural complexity

3.1.1 Overview of indices
A stand level index of structural complexity is a mathematical construct, which summarises the effects of two or more structural attributes, in a single number or index value. By acting as a summary variable for a larger pool of structural attributes, it is anticipated that, if properly designed, such an index could function as a reliable indicator of stand level biodiversity and provide a means of ranking stands in terms of their potential contribution to biodiversity (e.g. Parkes et al., 2003; Oliver et al., 2003; Neumann and Starlinger, 2001; Lahde et al., 1999; Van Den Meersschaut and Vandekerkhove, 1998; Koop et al., 1994). Some authors have erroneously used a diversity measure, such as the Shannon-Weiner index to quantify a single attribute and have then termed this attribute an index of structural complexity, when in fact they have quantified only one of many possible attributes (e.g. Gove et al., 1995; Buongiorno et al.,...
1994). In this review such measures are not treated as indices of structural complexity, and were discussed in Chapter 2.

Designing an index of structural complexity involves three key steps:
1. Selecting the number and type of attributes to be used in the index. This is not a trivial task because, as Chapter 2 has demonstrated, there are a wide variety of potential attributes.
2. Establishing the mathematical framework for combining attributes in a single index value.
3. Allocating a score or weighting to each attribute in the index.

There is little consensus in the literature as to how to approach these three steps, and few studies provide a clear rationale, other than the operation of expert opinion (e.g. Oliver, 2002; Parkes et al., 2003, Meersschaut and Vandekerkhove, 1998), for the selection of particular attributes in preference to others, or for the weighting of attributes when combined in an index. There is also a tendency for researchers to tailor indices to suit their immediate research needs (e.g. Watson et al., 2001; Newsome and Catling, 1979), available data (e.g. Acker et al., 1998), and forest type (eg Koop et al., 1994). As a result, the literature contains a variety of different indices with no single index preferred over the others. The most prominent of these indices are summarised in Table 2, and described in more detail in the following sections. For this purpose, indices have been grouped according to the mathematical framework that underpins the index.
Table 2: Indices used to quantify stand structural complexity.

<table>
<thead>
<tr>
<th>Index</th>
<th>Number of attributes</th>
<th>Comment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Structural Complexity Index (Barnet et al., 1978)</td>
<td>4</td>
<td>Additive index. Attributes describe small mammal habitat.</td>
</tr>
<tr>
<td>Habitat Complexity Score (Newsome and Catling, 1979; Watson et al., 2001&lt;sup&gt;a, b&lt;/sup&gt;)</td>
<td>5&lt;sup&gt;a, b&lt;/sup&gt;</td>
<td>Additive Index. Attributes describe small mammal habitat&lt;sup&gt;a&lt;/sup&gt;, or bird habitat&lt;sup&gt;b&lt;/sup&gt;.</td>
</tr>
<tr>
<td>Old-growth Index (Acker et al., 1998)</td>
<td>4</td>
<td>Measures degree of similarity to old-growth Douglas-fir conditions.</td>
</tr>
<tr>
<td>LLNS Diversity Index (Lahde et al., 1999)</td>
<td>8</td>
<td>Distinguishes successional stages of Finnish boreal forests.</td>
</tr>
<tr>
<td>Biodiversity Index (Van Den Meersschaut and Vandekerkhove, 1998)</td>
<td>18</td>
<td>Used to characterise biodiversity in Belgium forests. Attributes benchmarked against levels in forest reserves.</td>
</tr>
<tr>
<td>Vegetation Condition Score (Parkes et al., 2003&lt;sup&gt;c, d&lt;/sup&gt;; Oliver and Parkes, 2003&lt;sup&gt;e&lt;/sup&gt;; Gibbons et al., 2004&lt;sup&lt;f&lt;/sup&gt;)</td>
<td>8&lt;sup&gt;c, d, e&lt;/sup&gt;</td>
<td>Assesses vegetation condition in temperate Australian ecosystems. Attributes benchmarked at the scale of vegetation community.</td>
</tr>
<tr>
<td>Rapid Ecological Assessment Index (Koop et al., 1994)</td>
<td>9</td>
<td>Attribute levels benchmarked against levels in unlogged natural forest.</td>
</tr>
<tr>
<td>Extended Shannon-Weiner Index (Staudhammer and Lemay, 2001)</td>
<td>3</td>
<td>Uses an averaging system to extend the Shannon-Weiner Index to height, dbh and species.</td>
</tr>
<tr>
<td>Index of Structural Complexity (Holdridge, 1967; cited in Neumann and Starlinger, 2000)</td>
<td>4</td>
<td>Based on traditional stand parameters, which are multiplied together. Sensitive to number of species.</td>
</tr>
<tr>
<td>Stand Diversity Index (Jaehne and Dohrenbusch, 1997; cited in Neumann and Starlinger, 2000)</td>
<td>4</td>
<td>Combines measures for the variations in species, tree spacing, dbh and crown size.</td>
</tr>
<tr>
<td>Structural Complexity Index (Zenner, 2000)</td>
<td>2</td>
<td>Measures height variation based on tree height and spatial arrangement of trees.</td>
</tr>
<tr>
<td>Structure Index based on variance (STVI) (Staudhammer and Lemay, 2001)</td>
<td>2</td>
<td>Based on covariance of height and dbh. Independent of height or dbh classes.</td>
</tr>
</tbody>
</table>
3.1.2 Additive indices based on the cumulative score of attributes

This is the most straightforward means of constructing an index. A set of attributes is selected, with each attribute contributing a certain number of points to the index. Additive indices assume attributes are compensatory and independent, so that a reduction or absence of one attribute may be balanced by an increase in another (Burgman et al., 2001). For example if canopy cover and understorey vegetation cover were attributes in an additive index, then a reduction in canopy cover could be compensated by an increase in understorey cover. The value of an additive index is simply the sum of the scores of the attributes. In this approach the contribution of each attribute is easy to assess, and the final value of the index relatively simple to compute. However, the additive nature of the index can also mask important differences between stands. For example two stands can have the same index value, but this may be the result of quite different combinations of attribute scores (McCarthy et al., 2004).

One of the earliest and simplest additive indices was developed by Barnett et al. (1978) to incorporate the structural attributes important to Australian ground dwelling mammals into a single measure. They suggested an index of structural complexity based on four attributes, ground vegetation cover (<1m), shrub cover (1-2m), log cover, and litter cover. Attributes were assessed visually and then scored 0-3 on the basis of cover classes. Scores were then summed to give an index of structural complexity. The abundance of a variety of small mammal species was subsequently correlated with this index (Barnet et al., 1978).

Newsome and Catling (1979) extended this approach to include the attributes of tree canopy cover and soil moisture. Their index, or Habitat Complexity Score has also been correlated with the abundance of ground dwelling mammals (Catling et al., 2000; Catling and Burt, 1995), and in a modified form with bird species richness (Freudenberger, 1999). Habitat Complexity Score has also been suggested as a means of quantifying habitat heterogeneity. A large variance in Habitat Complexity Scores would be indicative of forests with high
levels of heterogeneity, whereas clumping of scores about a small variance would indicate a more uniform forest structure (Catling and Burt, 1995).

Acker et al. (1998) used an additive index to characterise Douglas-fir stands in Western Oregon and Washington. They termed their index an old-growth index \( I_{OG} \) because it benchmarked attributes relative to their mean value in old-growth stands. The index was based on four attributes considered by Spies and Franklin (1991) to successfully discriminate between age classes of Douglas-fir:

- Standard deviation of tree dbh
- Number of large (>100cm dbh) Douglas-fir trees
- Mean tree dbh
- Number of trees > 5cm dbh

Attributes describing dead wood (e.g. standing dead trees and logs), the density of shade-tolerant tree species, and the degree of layering in the forest canopy were not included in the index, despite Spies and Franklin (1991) having demonstrated their importance as structural attributes. This was because measurements of these attributes had not been made over the lifetime of the permanent plots used in the study. Of the four attributes used, each contributed 25% to the value of the index, which was computed as follows:

\[
I_{OG} = 25 \left[ \frac{(X_{i,obs} - X_{i,young})}{(X_{i,old} - X_{i,young})} \right]
\]

Where \( i = 1 \) to 4, representing each of the four structural variables, \( X_{i,obs} \) is the observed value of the \( i \)th structural variable, \( X_{i,young} \) is the mean value of the \( i \)th structural variable for young stands, and \( X_{i,old} \) is the mean value of the \( i \)th structural variable for old-growth stands. \( I_{OG} \) varies from 0 for a typical young stand, to 100 for a typical old-growth stand. Acker et al. (1998) successfully used the change in \( I_{OG} \) with time to quantify the rate of development of old-growth conditions in Douglas-fir forests.

Lahde et al. (1999) developed an additive index, called the LLNS Diversity Index, to characterise the structure of boreal forests in Finland. The authors considered variation in tree species and sizes, and the presence of dead standing and fallen trees to be key structural elements. They described these in terms of 8 attributes:

1. The size class distribution of different tree species, with larger size classes in
rarer species attracting a higher score;
2. The basal area of trees with dbh > 2cm;
3. The volume of standing dead trees;
4. The volume of fallen dead trees;
5. The density of seedlings;
6. The % cover of understorey plants;
7. The occurrence of special trees (rare because of their size or species);
8. The volume of charred wood with diameter > 10cm.

Attributes were quantified on the basis of classes (e.g. dbh class, volume class, density class), with different classes attracting different proportions of the total possible score allocated to the particular attribute. The value of the index was the sum of scores for each of the 8 attributes. Using data from the third National Forest Inventory of Finland, Lahde et al. (1999) found that their LLNS index distinguished between successional stages and site types of boreal forest more successfully than either the Shannon-Weiner or Simpson Indices of species diversity.

A more elaborate index was developed by Van Den Meersschaut and Vandekerkhove (1998) in order to characterise biodiversity within Belgium forests. They used 18 attributes in their index to describe elements of the overstorey, herb layer and dead wood, and also to reflect parameters considered likely to be affected by forest management. The selection and weighting of attributes were determined by a consensus of experts, and benchmark values for each attribute were based on an analysis of Belgium forest reserves judged most representative of the condition of natural forest stands. The maximum score for the index was 100, with points allocated to attributes as follows:

**Overstorey attributes (45);** canopy cover (4), stand age (7), number of canopy layers (4), number of tree species per unit area (5), number of native tree species (5), standard deviation of dbh (6), number of large trees (10), presence of natural regeneration (4).

**Herb layer composition (25);** richness of vascular plant species (10), degree of rareness (7), richness of bryophytes (5), total cover of herb layer (3).

**Dead wood (30);** basal area of stags (4), number of large trees (dbh>40cm) (6),
standard deviation of dbh of stags (5), total length of large logs (7), number of log diameter classes (8).

Van Den Meersehout and Vandekerkhove (1998) considered their index to successfully distinguish between a range of forest stands in Flanders, and to have ranked them in a logical order in terms of potential biodiversity value. This was partly attributed to the difference between the maximum and minimum index value, which was equivalent to 1/3 of the maximum score and left sufficient space to determine the biodiversity status of all the stands.

In Australia Parkes et al. (2003) used a similar approach to Van Den Meersehout and Vandekerkhove (1998) in the development of a vegetation quality index to quantify the habitat value of native vegetation. Their index is additive, and uses natural vegetation to benchmark values for the various attributes. However, unlike Van Den Meersehout and Vandekerkhove (1998) attributes are benchmarked at the scale of vegetation communities so that stands from different communities are assessed in terms of different benchmarks. The index also contains a landscape component, which accounts for 25% of the total score. The attributes and their weighting in the final index value of 100 are as follows:

**Stand structural complexity** (75%): assessed in terms of, large trees (10%), canopy cover (5%), abundance and richness of lifeforms in the understorey (25%), litter cover (5%), length of logs (5%), regeneration (10%) cover of weeds and weed species present (15%)

**Landscape context** (25%): assessed in terms of patch size (10%), proportion of landscape covered by neighbouring remnants (10%) distance to core area of habitat (5%).

McCarthy et al. (2004) question the validity of comparing attributes to a single benchmark state because many vegetation communities exhibit a variety of persistent states, rather than progressing to a single climax or equilibrium condition. This is particularly the case in the Australian environment where vegetation has often evolved in response to disturbance regimes associated with fire. As an alternative McCarthy et al. (2004) suggest a range of
benchmarks reflecting the different states produced by the particular disturbance regime. Parkes et al. (2003) partly address this source of variation by comparing attribute levels to a range (e.g. ± 50%) either side of their single benchmark value. The drawback with this approach is that this range can become so large that the index fails to differentiate between stands with quite different levels for a particular attribute.

Oliver and Parkes (2003) have proposed a modified version of the Parkes et al. (2003) index, as part of a toolkit for scoring the biodiversity benefits of land use change. In their version of the index the richness and cover of life forms are scored as separate attributes so that the cover provided by exotic vegetation can be included for its contribution to habitat. They also quantified the density of hollow bearing trees directly rather than assume a correlation between this attribute and the presence of large trees – an approach supported by McCarthy et al. (2004). The attributes and their weighting in the final index value of 100 are: richness of native plant groups (25), cover of all plant groups (20), regeneration (10), cover of weeds (15), litter cover (5), density of large trees (15), density of hollow bearing trees (5), length of logs (5).

### 3.1.3 Indices based on the average score of groups of attributes

An alternative to simply adding attributes to produce a final score is to find the average score of groups of attributes. Koop et al. (1994) used this approach to develop an index for the rapid ecological assessment of Sumatran rainforest. Attributes were placed in three groups, considered to characterise different elements of ecosystem integrity. The groups and their attributes were:

1. **Forest overstorey**: described by basal area, presence of large trees, maximum tree height, the number of distinct canopy layers, and the form of the diameter distribution (reverse J or other).
2. **Light transmission**: described by the abundance of pioneer species, the richness of light demanding species, and the richness of exotic invader species.
3. **Atmospheric moisture**: described by the presence of groups of species which indicate high humidity.

For each group attribute scores were tallied to give a score (D) which was
compared to benchmarks (R) established in unlogged forest. This allowed a relative score $S = (D/R) \times 100$, to be calculated for each group. The three relative scores were then averaged to give a final score. Koop et al. (1994) termed this index a measure of forest integrity because it compared attribute levels to those expected in a natural forest.

Staudhammer and Lemay (2001) used an averaging system to quantify three attributes (diameter class, height class and species) with the Shannon-Weiner Index ($H'$); where $H' = -\sum p_i \ln p_i$; and $p_i$ is the proportion of all individuals which occur in the $i$th species. To calculate $H'$ on the basis of diameter or height class $p_i$ was the proportion of stand basal area which occurred in the $i$th diameter or height class. Individual values for the Shannon-Weiner Index were calculated on the basis of height classes, dbh classes and species. These three index values were then averaged to give a final value reflecting all three attributes. Staudhammer and Lemay (2001) also applied the Shannon-Weiner index directly on the basis of height x dbh x species classes. Both approaches were judged successful in ranking a set of test stands in a logical order reflecting perceived biodiversity.

3.1.4 Indices based on the interaction of attributes

In this approach attributes are combined in an index in a non-linear fashion. The simplest method is to multiple attributes to give the final index value. In many situations multiplication will be inappropriate because it implies that structural complexity depends on the presence of all attributes (Burgman et al. 2001), since if one attribute has a zero value then the value of the index will also be zero. This situation can be addressed if attributes are limited to those that are concurrently present. Holdridge (1967, cited in Neumann and Starlinger, 2001) used this approach to combine traditional stand parameters in an index of structural complexity ($HC$) where

$$HC = H \times BA \times n \times N$$

$H$ is the top-height, $BA$ the basal area, $n$ the number of stems per ha, and $N$ the number of overstorey species. Neumann and Starlinger (2001) criticised this index on the basis that it is strongly influenced by the number of overstorey species and contained no information on within stand variation.
Jaehne and Dohrenbusch (1997 cited in Neumann and Starlinger 2001) partly address these issues by combining measures for the variations in species composition, diameter, tree spacing, and crown dimension in their Stand Diversity Index (SD), where:

$$SD = (\text{species variation}) \times (\text{dbh variation}) \times (\text{tree spacing variation}) \times (\text{crown variation})$$

Neumann and Starlinger (2001) found that HC and SD were both useful in characterising the structure of stands across a range of Austrian forest types. HC and SD were significantly correlated with each other and with the standard deviation of dbh. SD was also significantly correlated with overstorey species diversity.

Zenner (2000) constructed a Structural Complexity Index based on the interaction between tree height and the spatial location of trees. To do this, trees were represented as three dimensional data points, with the x, y coordinates representing horizontal position, and the z coordinate representing height. Groups of three adjacent points in this x, y, z space were connected to form a network of non-overlapping triangles. An index of tree height variation was then defined as the sum of the surface areas of these triangles divided by the horizontal area covered by the triangles. Zenner (2000) termed this index a Structural Complexity Index (SCI), although it quantified only two of many attributes of structure. The index equates increased structural complexity (higher index values) with increasing tree density and height variation. Canopy gaps are not recognised as increasing structural complexity, because these reduce the value of the index. The index has limited practical value because it requires the position and height of each tree to be precisely determined.

Finally, Staudhammer and Lemay (2001) have proposed an index based on the covariance of dbh and height (Structure index based on variance or STVI). The rationale for this index was that unlike the Shannon-Weiner index it would be independent of height or dbh classes. However the index is complex to compute, and reflects only two structural attributes. It was also the least preferred of the 4 indices tested by Staudhammer and Lemay (2001).
3.2 Conclusions

None of the indices described above provides a role model for developing an index of structural complexity. However, taken as a group the indices provide some useful guidance in approaching this task.

First, an index should be based on a comprehensive set of attributes. Relatively few indices currently do this. This largely reflects the arbitrary manner in which attributes have been selected. Most studies establish an attribute set by combining attributes the authors consider to be indicative of structure or biodiversity. How many attributes are included in this set appears to be a matter of subjective judgement, in which the number and type of attributes can vary considerably (see Table 2). The use of an alternative, “reductionist” approach could provide a more objective attribute set. In this approach a large initial set of attributes would be established using attributes with a demonstrated association with the elements of biodiversity that are of interest. This set could then be reduced to a core set by establishing correlations or other relationships between attributes.

Second, there are clear advantages in using a simple mathematical system to construct an index of structural complexity. This facilitates the use of multiple attributes and also makes it easier to visualise the output from the index in terms of real stand conditions. For example, compare the simple additive index of Van Den Meersschaut and Vandekerkhove (1998), which utilises 18 attributes, to the complex index developed by Zenner (2000) based on the interaction of just two attributes.

Third, structural attributes should be scored relative to the range of values occurring in stands of a comparable vegetation community (eg Parkes et al., 2003; Van Den Meersschaut and Vandekerkhove, 1998; Koop et al., 1994). The expected levels for structural attributes should therefore reflect the inherent characteristics of the site in question, and vegetation communities with naturally simple structures (e.g. single canopy layer with grassy understorey compared to multiple canopy layer with shrubby understorey) should have the potential to
achieve high scores on an index of structural complexity. This approach acknowledges that structural complexity is a relative, rather than absolute concept, and that uniformly high levels of structural attributes will not maximise biodiversity. This is because the presence of stands with naturally simple structures can increase the diversity of habitats in the landscape, and so contribute to beta diversity (Figure 1). There are also physiological reasons for this approach. For example, Specht and Specht (2002) indicate that the total projective foliage cover and biomass a stand can support is limited by climatic and edaphic factors. The potential levels of different structural attributes are therefore bound within certain limits, and the biota of that particular community will have evolved to reflect this range of variation.

As a final point, the weighting of attributes should be carefully considered as part of any index design. The literature provides little guidance as to how to do this, other than attempting to weight the contribution of attributes evenly (e.g. Watson et al., 2001; Acker et al., 1998; Koop et al., 1994). A minimum first step would be to test whether the operation of any proposed index is indeed sensitive to the weighting of attributes – for example by conducting a sensitivity analysis in which attributes are randomly weighted. If weighting does matter, then a range of weighting systems could be tested and the one that most clearly distinguishes between stands adopted.
Figure 1: Structural complexity is a relative, rather than absolute concept; illustrated here by three structurally complex stands from vegetation communities in southern Australia. (a) mixed forest Tasmania (photo P. Gibbons); (b) dry sclerophyll forest south-eastern NSW (photo C. McElhinny); (c) grassy woodland south-eastern NSW (photo P. Gibbons). In these stands the expected levels for structural attributes are different, and reflect the inherent characteristics of the physical environment in which each community occurs. The presence of stands (b,c) with naturally simple structures is important for biodiversity because they contribute to the variety of habitats in the landscape.
3.3 A methodology for developing an index of stand structural complexity

In light of the conclusions presented above, the following approach is proposed for developing an index of stand structural complexity:

1. Establish a comprehensive suite of stand structural attributes as a starting point for the index. Do this by reviewing studies in which there is an established relationship between elements of biodiversity and one or more structural attributes. I propose that an important source for this information will be fauna-habitat studies in which statistically significant relationships have been established between the presence or abundance of fauna and stand structural attributes.

2. Develop a measurement system for quantifying the many different attributes included in the comprehensive suite. Trial measurement techniques in a pilot study.

3. Use this measurement system to collect data from the vegetation communities in which the index is intended to operate. The communities should be sampled so that data are collected from a representative set of stands. These stands should reflect the range of vegetation condition (highly modified to unmodified) and developmental stages (regrowth to oldgrowth) occurring in each community.

4. Identify a core set of structural attributes from a univariate analysis of these data. For this purpose I propose that a core attribute should: a) function as a surrogate for other attributes through established correlations with these attributes, b) effectively distinguish between different stands as indicated by an even or approximately normal distribution of the attribute amongst study sites, and c) be efficient to measure and use in the field. Principal Components Analysis will be used to check for redundancy in the core set of attributes.

5. Combine the core attributes in a simple additive index, in which attributes are scored relative to their observed levels in each vegetation community. I propose to score attributes in the index using continuous functions rather than on the basis of arbitrary classes. Sensitivity analysis will be used to test the effect of weighting attributes in the index.
A flow chart summarising the five steps of the proposed methodology is shown in Figure 2.

To test the validity of this approach it was applied to woodland and dry sclerophyll communities in south-eastern Australia. In the next four chapters, I describe how this was done and what the results were. In Chapter 8, I compare the performance of the index that was developed from this process to other indices currently used in comparable Australian ecosystems.
Chapter 4: Fauna-habitat relationships, a basis for identifying key structural attributes in temperate Australian forests and woodlands.

Chapter summary
This chapter reviews a representative sample of the literature concerning fauna-habitat relationships in temperate Australian forests and woodlands as a basis for identifying some key structural attributes in these ecosystems. The review identifies fifty-five studies in south-east and south-west Australia in which the presence or abundance of different fauna were significantly (p<0.05) associated with vegetation structural attributes. The majority of these studies concerned bird, arboreal mammal, and ground mammal habitat requirements, with relatively fewer studies addressing the habitat requirements of reptiles, invertebrates, bats or amphibians. Thirty-four key structural attributes were identified from these fifty-five studies, by grouping similar attributes, and then representing each group with a single generic attribute. In combination with attributes identified in Chapter 2, the thirty-four attributes constitute a comprehensive set with a demonstrated association with biodiversity. This set provided the basis for developing an operational set of stand level attributes for the collection of data from study sites.

4.1 Introduction
In this chapter I address the first stage of the methodology for developing an index of structural complexity, which was proposed at the end of Chapter 3. The aim of this stage is to use a quantitative and objective process to identify a comprehensive suite of structural attributes. To do this, I review studies that have associated attributes of structure with the presence or abundance of fauna in Australian temperate forest and woodland ecosystems. The suite of attributes identified from this review forms the basis for subsequent stages of the methodology, which are addressed in chapters 5, 6 and 7.
4.2 The type of studies reviewed

I reviewed a representative sample of the literature and identified 55 studies in south-east and south-west Australia, which presented statistically significant (p<0.05) associations between the presence or abundance of different fauna, and vegetation structural attributes. Most of these associations involved direct correlations between attributes and the presence or abundance of fauna, although a few associations were in the form of significant effects in multivariate models.

The scope of the studies varied considerably; some, such as Fauna Impact Statements, investigated a range of different faunal groups while others focused on a single faunal species. Table 3 indicates the percentage of studies that focused on each of seven broad faunal groups. The majority of work concerned bird, arboreal mammal, and ground mammal habitat requirements, with relatively fewer studies addressing the habitat requirements of reptiles, invertebrates, bats or amphibians. In addition to these 55 quantitative studies, I also consulted a range of observational studies and reviews. While these were not used to identify structural attributes, they did provide additional information regarding the habitat requirements of the different faunal groups.

Table 3: The percentage of 55 studies that addressed seven broad faunal groups. Each study contained at least one statistically significant (p<0.05) association between the presence or abundance of fauna and vegetation structural attributes. Some studies addressed species from more than one faunal group, and hence sum to more than 100%.

<table>
<thead>
<tr>
<th></th>
<th>Birds</th>
<th>Arboreal mammals</th>
<th>Ground mammals</th>
<th>Bats</th>
<th>Reptiles</th>
<th>Amphibians</th>
<th>Invertebrates</th>
</tr>
</thead>
<tbody>
<tr>
<td>%</td>
<td>30</td>
<td>28</td>
<td>24</td>
<td>7</td>
<td>15</td>
<td>7</td>
<td>11</td>
</tr>
</tbody>
</table>

The results of the review are presented in two parts. In section 4.3 I discuss the habitat requirements and associated structural attributes for each faunal group. In section 4.4 I synthesise these attributes to produce a key set of generic attributes characterising the habitat requirements of all faunal groups.
Chapter 4: Fauna-habitat relationships, a basis for identifying key structural attributes

4.3 Habitat resources and structural attributes for seven faunal groups

4.3.1 Habitat resources used by birds

Overview

The bird species occurring in eucalypt woodlands and forests belong to a variety of foraging groups or guilds, reflecting the diversity of resources these vertebrates can utilise. A large proportion of bird species are insectivorous, foraging for invertebrates on the leaves, branches and bark of eucalypts, in the air spaces provided by canopy gaps, and amongst litter and woody debris (Woinarski et al., 1997). There is also a diversity of bird species that feed on nectar and exudates such as manna, honeydew and sap (Recher et al., 1985). A small proportion of birds feed on eucalypt seeds, and only a very few species are capable of digesting pollen. Frugivorous birds are rare and there are no leaf eating birds (Landsberg and Cork, 1997). In addition to food resources, birds utilise sites for nesting and shelter, either in the form of suitable tree hollows, or appropriate foliage or ground cover arrangements (Recher, 1991).

The most productive habitats for birds appear to be those providing a range of resources that can support a variety of different foraging groups (Bauer et al., 2000). This is most likely to occur where there is a diversity of vegetation strata available as foraging substrates (Gilmore, 1985; Loyn, 1985a; Recher, 1969). In Australian temperate forests and woodlands, different strata tend to reflect differences at the levels of plant life form (forb, shrub, tree etc.) and plant genus. This provides different kinds of food (nectar, fruit, seeds) and foliage thereby increasing the diversity of foraging opportunities for bird species (Recher, 1985). Six habitat components have been consistently identified as important resource bases for birds:

- **Foliage**: a source of exudates and invertebrates;
- **Flowers**: a source of nectar and invertebrates;
- **Bark**: a source of exudates and invertebrates;
- **The ground layer**, including ground vegetation, litter, logs and coarse woody debris; - a source of invertebrates and small vertebrates;
- **Air spaces**: within and between canopy strata a source of invertebrates;
- **Hollow-bearing trees**: for nesting and shelter.

A variety of structural attributes have been used to characterise these habitat components. These attributes and the literature relevant to each habitat component are discussed below. As a summary, those structural attributes significantly associated with the presence, abundance or diversity of bird species or assemblages of species are listed in Table 4.

Table 4: Structural attributes significantly (p<0.05), and positively associated with the presence, abundance or richness of one or more bird species in eucalypt forests (F) or woodlands (W). (-) indicates a study reporting a significant negative association for some species.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Foliage Height Diversity</td>
<td>(W) Abbott 1976, Recher 1969</td>
</tr>
<tr>
<td>Canopy height</td>
<td>(F)(W) Smith et al. 1994, Gilmore 1985</td>
</tr>
<tr>
<td>Canopy surface area and volume</td>
<td>(F)(W) Gilmore 1985</td>
</tr>
<tr>
<td>Number of overstorey stems</td>
<td>(F)(W) Pearce 1996, Arnold et al. 1987</td>
</tr>
<tr>
<td>Basal area of overstorey stems</td>
<td>(F) Braithwaite et al. 1989</td>
</tr>
<tr>
<td>Overstorey species richness</td>
<td>(F) Bauer et al. 2000, Smith et al. 1994</td>
</tr>
<tr>
<td>Foliar nutrients (Magnesium index)</td>
<td>(F) Braithwaite et al. 1989</td>
</tr>
<tr>
<td>Flowering intensity index</td>
<td>(F) McGoldrick and MacNally 1998</td>
</tr>
<tr>
<td>Bark index</td>
<td>(F)(W) Pearce 1996, Gilmore 1985</td>
</tr>
<tr>
<td>% cover midstorey trees</td>
<td>(F) Fanning 1995</td>
</tr>
<tr>
<td>Shrub species richness</td>
<td>(W) Seddon et al. 2001, Arnold et al. 1987,</td>
</tr>
<tr>
<td>Mistletoe density</td>
<td>(W) Watson 2002</td>
</tr>
<tr>
<td>% cover or other measure of litter abundance</td>
<td>(F)(W) Watson 1999*, Smith et al. 1994, Gilmore 1985</td>
</tr>
<tr>
<td>% cover bare soil</td>
<td>(F) Smith et al. 1994 (-)</td>
</tr>
</tbody>
</table>

* As component of Habitat Complexity Score, attribute was correlated with bird species richness
Bird habitat components

FOLIAGE

Foliage carbohydrate sources such as manna, lerp and honeydew form an important component of the diet of a wide range of bird species (Woinarski et al., 1997; Recher et al., 1985). These resources are more abundant on eucalypt rather than on non-eucalypt plant species, with lerp producing insects being most abundant on eucalypts with high foliar nutrient content (Recher, 1991). Attributes that are indirect measures of these carbohydrate resources are the basal area of eucalypt species, and foliar magnesium, both of which have been correlated with bird species richness (Braithwaite et al., 1989).

The distribution and abundance of bird species have also been related to particular eucalypt subgenera. Loyn (1985b) found that there were differences in the bird species associated with eucalypts from the *Symphyomyrtus* and *Monocalyptus* subgenera. This was attributed to differences in the invertebrates which occurred on the foliage of the two subgenera, and to the relative abundance of hollow-bearing trees in the *Symphyomyrtus* subgenus—although Gibbons (1999), found there were no significant differences between these subgenera in the rate of occupancy of hollows by fauna. Recher (1985) attributed an increased abundance and diversity of bird species in forests dominated by *Symphyomyrtus* to the typically higher foliage nutrient levels of this subgenus compared with the *Monocalyptus* subgenus.

Non-eucalypt species are also important to many foliage-gleaning insectivorous birds because these plants offer a greater diversity of foliage arrangements and associated invertebrates than eucalypts (Woinarski et al., 1997). Shrub cover and shrub species richness is a direct measure of this resource and has been repeatedly associated with bird species richness (Seddon et al., 2003; Seddon et al., 2001; Andrews et al., 1994; Arnold et al., 1987; Recher et al., 1985).

Although there is relatively little foraging height specialisation amongst eucalypt and woodland birds (Recher et al., 1985), some bird assemblages have been shown to favour the foliage available in different vegetation strata. In a study of
vegetation remnants in Western Australia, Arnold et al. (1987) found that abundance of the raven (*Corvus coronoides*), magpie (*Gymnorhina tibicen*), mudlark (*Grallina Cyanoleuca*) and galah (*Cacatua roseicapilla*) increased with greater tree canopy cover and less shrub cover, while the number of small insectivorous passerines increased as tree canopy decreased and shrub cover and shrub species increased.

The presence of different vegetation strata may also impact on the breeding success of birds. Most eucalypt forest and woodland birds nest above the ground and within a specific layer of vegetation, placing their nests among the foliage and small branches of trees and shrubs. The choice of nest site is often related to foliage density and plant species so that each bird species has a limited capacity to adjust its nesting height. Habitats with sparse foliage will usually support relatively few nest sites compared with habitats with a complex vegetation profile and dense foliage (Recher, 1991). For example, the presence of mistletoe clumps provides dense foliage that is favoured for nesting sites by some bird species, so that in some Australian woodlands mistletoe density can have a significant positive effect on bird species richness (Watson, 2002). Gilmore (1985) found that canopy height was an effective and simple measure of the diversity of vegetation strata, and that this could be used to predict the density of insectivorous birds in a range of eucalypt and acacia woodlands in Victoria and central Queensland.

**FLOWERS – A TEMPORAL HABITAT COMPONENT**

Birds can be highly mobile and often utilise different habitats at different times of the year (Reid, 1999). For example many nectarivorous birds follow the asynchronous flowering of eucalypt and understorey species, locating and exploiting sources of nectar production as they occur in different woodland and forest communities (McGoldrick and MacNally, 1998; Traill, 1993). To facilitate this, birds often nest in high quality sites from late winter to summer when abundant insect populations and foliage production provide the density of resources needed for reproduction. The birds are then free to either migrate, or to locate winter flowering eucalypts and shrubs elsewhere in the landscape (Loyn, 1993; Recher, 1985; Loyn, 1985).
The timing and abundance of these floral resources has been related to
overstorey species composition (MacNally and McGoldrick, 1997; Kavanagh,
1984), the floristic diversity of the understorey (Pyke, 1985; Recher, 1985), and
the presence of key species, which flower profusely (Watson, 2002; Turner,
1991), or at a critical time (Reid, 1999; Loyn, 1985). Floral food resources have
also been quantified using indices of flowering intensity, and these indices have
been correlated with the abundance and richness of nectarivores (McGoldrick
and MacNally, 1998; Ford, 1983).

Simple attributes such as shrub richness and cover, mistletoe density and the
distribution of basal area amongst overstorey species may also effectively
describe the potential floral resource.

**Bark**

Bark is an important habitat component in eucalypt forests and woodlands and
is used to some degree by nearly all bird species. In some habitats bark can
account for up to 25% of all foraging manoeuvres (Recher et al., 1985). Bark
foraging birds utilise a range of invertebrates and sugar rich exudates found on
and behind decorticating and non-decorticating bark (Recher and Davis, 1998;
Loyn, 1985). Decorticating bark may also provide perching sites, which some
birds require to access manna and to probe for arthropods (Pearce et al.,
1994).

Noske (1985) indicated that variation in bark structure and bark type influences
the distribution of bark-dwelling invertebrates and the foraging tactics of birds
preying upon them. A number of other studies also conclude that birds are likely
to be locally diverse in association with the contrasting bark characteristics of
mixed eucalypt communities (Woinarski et al., 1997; Landsberg and Cork,
1997; Recher, 1991). Indices that quantify the availability of decorticating bark
have been correlated with the presence of insectivorous birds (Pearce, 1996;
Gilmore, 1985). However, no studies appear to have quantified the importance
of a variety of bark types for bark foraging birds.
THE GROUND LAYER

Ground foraging birds, which utilise ground vegetation (e.g. grass, herbs), litter, logs and coarse woody debris as foraging substrates are one of the most abundant bird communities in the woodlands and forests of south-western and south-eastern Australia (Recher and Davis, 1998). This reflects the absence of dense ground vegetation in many woodlands and dry sclerophyll forests, and the accumulation of litter and woody debris in wet and dry sclerophyll forests (Bauer et al., 2000; Recher and Davis, 1998; Woinarski et al., 1997). Many threatened and declining bird species in western NSW are ground foraging insectivorous birds and this has been partly attributed to the loss of a structurally diverse ground layer with abundant litter and coarse woody debris (Reid, 1999).

Litter and woody debris are relatively straightforward to quantify, and both have been used to characterise the value of the ground layer for birds. Smith et al., (1994) described the litter layer in terms of percentage litter cover, and showed that this was positively correlated with the richness of ground foraging bird species and negatively correlated with the cover of grasses and herbs. However, litter cover only partially describes the resource available for ground foragers because the depth of the litter layer will vary within and between habitats. The dry weight of litter may therefore be a better measure of available organic matter and this has been correlated with the density of ground foraging birds (Gilmore, 1985).

Relatively few studies have quantified the role of fallen timber in the provision of bird habitat, although it appears that woody debris will provide significant food and foraging opportunities, and shelter sites (Reid, 1999; Recher and Davis, 1998). Laven and MacNally (1998) found that piles of fallen timber influenced the spatial location of birds in Victorian Box-Ironbark forests. Bird numbers were nine times greater, and bird species three times richer, in areas containing piles of woody debris.

CANOPY AIR SPACES

The air spaces within and between canopy layers are an important source of
invertebrates for insectivorous birds (Landsberg and Cork, 1997; Recher, 1991). In one of the few studies that have attempted to quantify this resource, Gilmore (1985) correlated the density of aerially foraging insectivores with an index of tree crown surface area. However, this index required measurements of individual tree crowns, and Gilmore concluded that canopy height was the simplest estimator of insectivore densities.

HOLLOWS

Of the 777 bird species in Australia, 15% (114 species) utilise hollows (Gibbons and Lindenmayer, 2002), and 7% (57 species) were considered by Saunders et al. (1982) to be obligate hollow users. For many of these species the availability of hollows can determine their presence or absence from a given site (MacNally et al., 2000, Traill, 1991). Several authors have found a significant relationship between the density of tree hollows and the diversity of forest or woodland bird species (Seddon et al., 2003; Seddon et al., 2001; Fanning, 1995). However, this relationship has not been replicated for some hollow-dependent species such as owls, parrots, rosellas, and cockatoos. These species can travel long distances and consequently the number of hollows measured at a particular site may not reflect the availability of hollows over a larger area (Smith et al., 1994; Andrews et al., 1994).

Choice of nest hollow characteristics can be highly specific with different bird species preferring different entrance sizes and hollow depths (Gibbons and Lindenmayer, 2002). Recher (1991) observed some bird species selecting smooth rather than rough openings to the hollow, or dead rather than living trees, or one tree species over another. Competition for hollows can occur between bird species, which utilise the same type of hollow, and in this situation dominant species may exclude other species by occupying all suitable hollows (Newton, 1994). A diversity of hollow types rather than an abundance of a few types, is therefore more likely to support a diversity of bird species.

4.3.2 Habitat resources used by arboreal mammals

Overview

The distribution and abundance of arboreal mammals in the landscape is often
patchy (Braithwaite et al., 1983), reflecting an association with high quality and floristically diverse areas of forest (Pausas et al., 1995; Davey, 1984). These areas are preferred because they are likely to provide year round resources at a scale comparable with the home ranges of arboreal mammals (Eyre and Smith, 1997; MacFarlane, 1988; Kavanagh, 1987; Kavanagh, 1984). Trees that flower or shed their bark in winter are particularly important, and where these are missing, animals are also absent or only present in low numbers (Soderquist and MacNally, 2000; Kavanagh, 1987).

The following four habitat components appear to provide critical resources for arboreal mammals:

- **Foliage**: a source of edible leaf material;
- **Flowers**: a source of nectar and pollen;
- **Bark**: a source of exudates and invertebrates;
- **Hollows**: as den and nesting sites and as sources of free water.

A variety of structural attributes have been used to characterise these habitat components and those attributes significantly associated with the presence, abundance or richness of arboreal mammals are listed in Table 5.

**Arboreal mammal habitat components**

**FOLIAGE**

Many studies have demonstrated the importance of foliar nutrient concentration in controlling the distribution and abundance of arboreal mammals in eucalypt forests and woodlands (Munks et al., 1996; Pausas et al., 1995; Kavanagh and Lambert, 1990; Braithwaite et al., 1983). At a micro-scale, animals appear to preferentially browse on young leaves of particular tree species so as to maximise their intake of foliar nitrogen and minimise the bulk of indigestible fibre (Kavanagh and Lambert, 1990). At a macro-scale animals are associated with high quality sites that are floristically diverse, because this increases the choice of food sources, so that at any one time the most nutritious foliage can be selected (Pausas et al., 1995; Braithwaite et al., 1983).
Table 5: Structural attributes significantly (p<0.05), and positively associated with the presence, abundance or richness of one or more arboreal mammal species in eucalypt forests (F) or woodlands (W). (-) indicates a study reporting a significant negative association for some species.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Basal area of overstorey stems</td>
<td>(F)(W) Munks <em>et al.</em> 1996, Braithwaite <em>et al.</em> 1983,</td>
</tr>
<tr>
<td>Number or basal area of large trees (&gt;60cm(^A),&gt;80cm(^B), other(^C)) used directly or as part of an index</td>
<td>(F) Soderquist and MacNally 2000(^{BC}), Pausas <em>et al.</em> 1995(^{A}), Andrews <em>et al.</em> 1994(^{B}), Kavanagh and Lambert 1990(^{B}), Hindell and Lee 1987(^{C}), Braithwaite <em>et al.</em> 1983(^{AB})</td>
</tr>
<tr>
<td>Overstorey diversity (Shannon Weaver Index(^D), species richness(^E))</td>
<td>(F)(W) Braithwaite <em>et al.</em> 1983(^{D}), Munks <em>et al.</em> 1996(^{E})</td>
</tr>
<tr>
<td>Canopy height (Site height(^F))</td>
<td>(F) Soderquist and MacNally 2000, Eyre and Smith 1997(^{F}), Smith <em>et al.</em> 1994</td>
</tr>
<tr>
<td>Foliage density at different heights or strata</td>
<td>(F) Jackson 2000, Davey 1984</td>
</tr>
<tr>
<td>Number of hollow-bearing trees</td>
<td>(F) Smith <em>et al.</em> 1994, Lindenmayer <em>et al.</em> 1991a, Lindenmayer <em>et al.</em> 1990b</td>
</tr>
<tr>
<td>Spatial distribution of hollow-bearing trees</td>
<td>(F) Lindenmayer <em>et al.</em> 1990a</td>
</tr>
<tr>
<td>Measures of foliar nutrients (leaf water content(^G), Foliage Nutrient Index(^H), BA of low foliar nutrient eucalypt species(^I))</td>
<td>(F)(W) Munks <em>et al.</em> 1996(^{G}), Pausas <em>et al.</em> 1995(^{H}), Kavanagh and Lambert 1990, Braithwaite <em>et al.</em> 1983(^{I}) (-)</td>
</tr>
<tr>
<td>Eucalypt species richness</td>
<td>(F) Bauer <em>et al.</em> 2000</td>
</tr>
<tr>
<td>% cover canopy trees</td>
<td>(F) Andrews <em>et al.</em> 1994</td>
</tr>
<tr>
<td>% cover shrubs or other abundance measure</td>
<td>(F) Andrews <em>et al.</em> 1994, Lindenmayer <em>et al.</em> 1991a, Lindenmayer <em>et al.</em> 1990b</td>
</tr>
<tr>
<td>Flowering Index</td>
<td>(F) Kavanagh 1987</td>
</tr>
<tr>
<td>Bark index</td>
<td>(F) Pausas <em>et al.</em> 1995, Lindenmayer <em>et al.</em> 1990a, Kavanagh 1987,</td>
</tr>
</tbody>
</table>

In light of these preferences a critical threshold in foliage nutrients for arboreal mammal folivores has been suggested, with forests whose leaves fall below this threshold unable to sustain viable populations of animals (Cork and Catling, 1996; Pausas *et al.*, 1995). Such a threshold may explain the concentration of 63% of arboreal mammals occurring in the Eden Management Area of south-
east NSW, in only 9% of the forested area (Cork and Catling, 1996; Braithwaite et al., 1983).

In addition to the quality of foliage, its vertical arrangement also affects the abundance and diversity of arboreal mammal folivores. Different species have been shown to utilise different strata within the canopy and to prefer different densities of foliage (Jackson, 2000; Smith et al., 1994; MacFarlane, 1988; Davey, 1984). Vertical complexity is likely to be associated with high levels of floristic diversity, since this provides a range of plants with different foliage densities, height development and shade tolerance (Florence, 1996; Davey, 1984).

Simple attributes, which have been used to quantify the foliage component of arboreal habitat, include stand basal area, canopy height and overstorey species composition. Stand basal area and canopy height are simple measures of site fertility, canopy biomass and vertical complexity (Carron, 1968 cited in Davey, 1984), and both have been correlated with the abundance of arboreal mammals (Eyre and Smith, 1997; Smith et al., 1994; Braithwaite et al., 1983). Overstorey species richness is a direct measure of floristic diversity and has also been correlated with the abundance and diversity of arboreal mammals (Bauer et al., 2000; Braithwaite et al., 1983).

FLOWERING

The distribution and abundance of flowering plants, particularly those flowering in winter, has been shown to be critical for a range of arboreal species (Soderquist and MacNally, 2000; Smith et al., 1994; Andrews et al., 1994; Kavanagh, 1987). Although indices have been developed to quantify the resources provided by flowers (Kavanagh, 1987), in practice these can be complex to use. Quantifying the temporal availability of flowering resources is also difficult because it requires multiple measurements over successive time periods. Since the timing and abundance of flowering depends largely on the species present and the size of the trees (Smith et al., 1994; Kavanagh, 1987), attributes such as distribution of basal area amongst overstorey species and mean dbh may provide a simple and more practical means of characterising the
availability of floral resources.

**Bark**

Bark is an important resource for arboreal mammals because many species forage for invertebrates, which shelter under decorticating bark (Eyre and Smith, 1997; Lindenmayer *et al.*, 1990a). These invertebrates provide an important source of protein in the diet of many arboreal mammals. Large numbers of invertebrates over-winter under the bark of eucalypts, so that tree species, which shed their bark in winter, increase the availability of this food resource at a critical time of the year (Kavanagh, 1984). Large gum-barked trees may be particularly important because their large surface area of decorticating bark provides a concentrated source of invertebrates and exudates (Eyre and Smith, 1997). The presence and abundance of a range of arboreal species have been correlated with indices which quantify the amount of decorticating bark (Pausas *et al.*, 1995; Lindenmayer *et al.*, 1990a; Kavanagh, 1987). Simpler attributes also likely to characterise bark resources are the distribution of basal area amongst overstorey species, and mean dbh.

In addition to invertebrates, sap exudates present on the bark of acacia understorey species provide important carbohydrate sources for arboreal mammals. The basal area of acacia species is an indirect measure of this resource and has been correlated with the abundance and diversity of arboreal mammals in the Mountain Ash forests of Victoria (Lindenmayer *et al.*, 1991a).

**Hollows**

There are 40 species of arboreal and scansorial mammals that utilise tree hollows in Australian forests and woodlands (Gibbons and Lindenmayer, 2002). This equates to approximately 13% of the 303 species of Australian terrestrial vertebrates that use hollows. The presence, abundance and diversity of arboreal mammals have been correlated with the number of hollow-bearing trees (Lindenmayer and Franklin, 1997; Gibbons and Lindenmayer, 1996; Smith *et al.*, 1994). Low numbers of hollow-bearing trees have also been shown to limit densities of arboreal mammals in both forests and woodlands (Bauer *et al.*, 2000; Lindenmayer *et al.*, 1999; Lindenmayer *et al.*, 1991a; Traill, 1991; Lindenmayer *et al.*, 1990b).
Diameter at breast height (dbh) is the most common attribute used to describe the availability of hollow-bearing trees for fauna (Wormington et al., 2003; Gibbons et al., 2000; Shelley, 1998; Pausas et al., 1995; Bennet et al., 1994; Lindenmayer et al., 1991b; MacKowski, 1984). High basal area, which often reflects a mature stand, may also be indicative of the presence of hollow-bearing trees (Braithwaite et al., 1983) although Gibbons et al., (2000) only found a weak association between basal area and the density of hollow-bearing trees. In a review of 13 different studies, which addressed the occupancy of trees by hollow dependent fauna, Gibbons et al., (2002), found that 80% of studies reported a significant correlation between occupancy and tree dbh. As an attribute, dbh should include dead as well as live trees, because dead trees may account for up to 18.5% of all trees with hollows, and are preferred by some arboreal vertebrate species (Gibbons et al., 2002).

While trees with larger dbh are likely to contain more occupied hollows, Gibbons et al., (2002) found that these were predominantly large hollows, with relatively few small, occupied hollows, and that greatest numbers of hollow-bearing trees occurred within the mid-range of diameter classes. Different arboreal species utilise different sized hollows, both in terms of preferred entry size and internal hollow dimensions (Whitford, 2001; Lindenmayer and Franklin, 1997; Gibbons and Lindenmayer, 1996), so that maximum diversity of arboreal fauna will require a range of hollow-bearing tree sizes. Overstorey species composition could be an attribute associated with a diversity of hollow types, because eucalypt species can develop different types of cavity as a result of differences in growth form and morphology, and in their susceptibility to attack by decay organisms (Wormington et al., 2003; Lindenmayer et al., 2000b). On this basis, the potential hollow resource for arboreal species is better described either by direct observation of hollows in different entry size classes, or by the average dbh of living and dead trees, its coefficient of variation, and overstorey species composition, rather than by tree dbh alone.

In addition to these attributes, the spatial distribution of hollow-bearing trees is an important consideration. Lindenmayer et al., (1990a) found that the territorial behaviour of some arboreal species limited the availability of hollows when
trees were clustered rather than more uniformly spaced. The spacing of hollow-bearing trees has been quantified using the Cox Index of clumping, which is the variance to mean ratio of the number of hollow-bearing trees per unit area (Lindenmayer et al., 1990a).

### 4.3.3 Habitat resources used by ground mammals

**Overview**

Other things being equal, habitat quality for ground mammals appears to be primarily determined by understorey complexity (Cork and Catling, 1996). Four habitat components have been commonly used to characterise the understorey resources which are important to ground mammals:

- **Shrub cover**: for shelter and refuge from predators;
- **Logs and coarse woody debris**: for shelter and nesting and as a source of invertebrates and edible fungi;
- **Litter**: for nesting and as a source of invertebrates;
- **Rocks**: for refuge and den sites.

A variety of structural attributes have been used to quantify these habitat components and those attributes significantly associated with the presence, abundance or richness of one or more species of ground mammal are listed in Table 6.

Different groups of ground mammal species are often associated with different levels of complexity in the arrangement of the four structural components listed above (Cork and Catling, 1996; Catling and Burt, 1995; Barnet et al., 1978). Large mammals such as the eastern grey kangaroo (*Macropus giganteus*) and the common wombat (*Vombatus ursinus*) prefer an open understorey with scattered shrubs and a ground cover of grasses and are likely to increase in abundance as understorey complexity decreases (Catling and Burt, 1995); whereas small to medium sized ground mammals such as antechinus, potoroos and bandicoots prefer a complex understorey having a diverse shrub layer and a ground cover of logs, leaf litter and woody debris (Paull and Date, 1999).
Table 6: Structural attributes significantly (p<0.05), and positively associated with the presence, abundance or richness of one or more ground mammal species in eucalypt forests. (-) indicates a study reporting a negative association for some species. There were no relevant studies for woodlands.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Basal area of overstorey stems</td>
<td>Catling <em>et al.</em>, 2000, Catling and Burt, 1995</td>
</tr>
<tr>
<td>Shrub and overstorey species richness</td>
<td>Bennett, 1993</td>
</tr>
<tr>
<td>% midstorey cover</td>
<td>Andrews <em>et al.</em>, 1994</td>
</tr>
<tr>
<td>Vertical density of shrub layer foliage</td>
<td>Bennett, 1993</td>
</tr>
<tr>
<td>% cover of debris (litter, logs and rocks)</td>
<td>Catling <em>et al.</em>, 2000*, Catling and Burt, 1995*</td>
</tr>
</tbody>
</table>

*Attribute a component of Habitat Complexity Score, which was correlated with the presence and abundance of ground mammals.

Some ground mammals require mosaics or ecotones that allow the inclusion of contrasting microhabitats within their home ranges (Smith *et al.*, 1994). Long-nosed potoroos (*Potorous tridactylus*) for example, utilise a range of vegetation densities – thick cover for shelter and protection from predators and open areas to forage for hypogeal fungi (Claridge and Barry, 2000; Bennett, 1993).

The utilisation of a range of vegetation structures by ground mammals implies that maximum diversity of ground mammals is likely to occur in habitats comprising a small-scale mosaic of dense and open vegetation patches (Lunney and Ashby, 1987). This type of heterogeneity could be quantified by coefficients of variation in structural attributes such as basal area of overstorey species, tree dbh, and percentage cover of overstorey and understorey species. However, to date such measures have not been used to characterise the habitat of ground mammals (Claridge and Barry, 2000).
Ground mammal habitat components

SHRUB COVER
Understorey vegetation provides shelter and cover for a wide range of ground mammals. Medium to large ground mammals such as the eastern grey kangaroo and common wombat utilise patches of dense vegetation to provide cover adjacent to open habitats with a grassy understorey for grazing (Catling et al., 2000; Lunney and O’Connell, 1988). Small ground mammals such as the brown antechinus (Antechinus stuartii), yellow-footed antechinus (Antechinus flavipes) and fawn-footed melomys (Melomys cervinipes) have a preference for densely vegetated habitats with a well developed shrub layer in which they can hide and forage for invertebrates (Catling et al., 2000; Catling and Burt, 1995; Smith et al., 1994; Bennett, 1993).

Attributes such as percentage cover of shrubs, the vertical density of shrub foliage, and the floristic richness of the shrub layer have been used to quantify shrub cover, and each has been associated with the presence or abundance of a range of small, medium or large ground mammals (Catling et al., 2000; Claridge and Barry, 2000; Catling and Burt, 1995; Smith et al., 1994; Bennett, 1993). The abundance of medium and large mammals has also been correlated with the basal area of overstorey species. This reflects a decline in shrub cover associated with an increasing basal area of the overstorey (Catling et al., 2000; Catling et al., 1998; Catling and Burt, 1995)

LOGS AND WOODY DEBRIS
Logs are a critical resource for small ground mammals. They provide shelter from weather, refuge from predators (Woinarski et al., 1997), nesting sites for lactating females (Woodgate et al., 1994; Smith et al., 1989), and in seasonally cold climates logs reduce energy expenditure for hibernating species such as the echidna (Tachyglossus aculeatus) (Smith et al., 1989). Logs are also important for the mobility of small ground mammals, providing easily traversed travel routes through dense undergrowth (Halstead-Smith, 1999 cited in MacNally et al., 2001; Woodgate et al., 1994). A variety of log sizes and conditions are utilised by small ground mammals, ranging from dry intact hollow logs for nest sites (Smith et al., 1989), to decomposing moist logs which provide
a substrate for food resources such as hypogaeal fungi and invertebrates (Woodgate et al., 1994; Dickman, 1991). Mature trees are indirectly a critical resource for many small ground mammals because these trees are the future source of hollow logs (Smith et al., 1994; Andrews et al., 1994; Dickman, 1991). The richness of small ground mammal species has been correlated with the abundance of large logs (Bauer et al., 2000; Smith et al., 1994; Andrews et al., 1994), as has been the density of individual species such as the yellow-footed antechinus (MacNally et al., 2001).

LITTER
Some small ground mammals utilise a deep litter layer in which to burrow for shelter and to forage for invertebrates (Dickman, 1991; MacFarlane, 1988). Percentage cover of litter is the attribute commonly used to quantify this resource, and has been correlated with the abundance of a number of small ground mammals including brown antechinus and fawn-footed melomys (Barnett et al., 1978).

ROCKS
Rocky areas can provide shelter and refuge sites for a range of ground mammals including echidna, common wallaroo (Macropus robustus), brush-tailed rock wallaby (Petrogale penicillata), and tiger quoll (Dasyurus maculatus) (Paull and Date, 1999). However, no studies appear to have quantified the value of this resource, or to have correlated it with the presence or abundance of ground mammals. Newsome and Catling (1979) incorporated rock cover into a measure of ground cover, which also included logs, woody debris and leaf litter. This measure formed part of an index of habitat complexity, which has been correlated with the presence and abundance of ground mammals (Catling et al., 2000; Catling and Burt, 1995).

4.3.4 Habitat resources used by bats

Overview
Bats comprise more than 25% of mammal species in Australia (Smith et al., 1994). Their diversity is largely determined by foliage structure, with many bat species exhibiting distinct foraging preferences for particular vegetation layers.
Chapter 4: Fauna-habitat relationships, a basis for identifying key structural attributes

(Brown et al., 1997; Andrews et al., 1994; O’Neill and Taylor, 1986). Species with large wing areas and small mass are slow flying yet highly manoeuvrable, and prefer to forage in dense forest layers; whereas large fast flying species are less manoeuvrable and prefer open spaces above forests or within mid-storey gaps below tree crowns (Brown et al., 1997; Andrews et al., 1994). The richest bat assemblages tend to occur on high productivity sites with well-developed vegetation strata (Smith et al., 1994).

Three habitat components appear to determine the presence and abundance of bat species:

- **Foliage and canopy spaces**: for foraging;
- **Hollows and decorticating bark**: for roost and nesting sites;
- **Permanent water**.

The structural attributes used to quantify these habitat components are listed in Table 7. The individual habitat components are then discussed below.

Table 7: Structural attributes significantly (p<0.05), and positively associated with the presence, abundance or richness of one or more species of bats in eucalypt forests. (-) indicates a study which reported a negative association for some species. There were no relevant studies for woodlands.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Overstorey dbh</td>
<td>Lunney et al., 1988</td>
</tr>
<tr>
<td>Canopy height</td>
<td>Brown et al., 1997</td>
</tr>
<tr>
<td>Diameter distribution amongst overstorey species</td>
<td>Lunney et al., 1988</td>
</tr>
<tr>
<td>Height difference between canopy and midstorey</td>
<td>Brown et al., 1997</td>
</tr>
<tr>
<td>Midstorey height</td>
<td>Brown et al., 1997</td>
</tr>
<tr>
<td>% cover of midstorey</td>
<td>Andrews et al., 1994 (-)</td>
</tr>
<tr>
<td>% cover of shrubs</td>
<td>Smith et al., 1994</td>
</tr>
<tr>
<td>% cover of litter</td>
<td>Andrews et al., 1994 (-)</td>
</tr>
<tr>
<td>% cover of grass</td>
<td>Andrews et al., 1994</td>
</tr>
<tr>
<td>Maximum dbh unburnt trees</td>
<td>Andrews et al., 1994</td>
</tr>
</tbody>
</table>

3.3.4.2 Bat habitat components

**FOLIAGE AND CANOPY SPACES**

Insectivorous bats forage on and amongst plant foliage, and in the air spaces between different vegetation layers. O’Neill and Taylor (1986) identified four
distinct foraging patterns in Tasmanian bat species:

- Slow highly manoeuvrable species, which foraged close to ground vegetation and the shrub layer;
- Faster manoeuvrable species, which foraged in and above the shrub layer;
- Fast but less agile species, which foraged in the space between the top of the understorey and canopy;
- Fastest but least agile species, which foraged above or just below the overstorey.

O’Neill and Taylor (1986) concluded that maximum diversity of bat species would require a patchy canopy with well-developed shrub and subcanopy strata.

Brown et al., (1997) found that insectivorous bat activity increased with stand age in montane ash forests in central Victoria, reflecting increased vertical separation between the primary and secondary strata. The space between these two strata also became less cluttered as the number of overstorey stems declined. Bat activity was significantly correlated with the height of the primary tree stratum, the height of the secondary tree stratum and the height difference between these two strata.

HOLLOW-BEARING TREES

Hollow-bearing trees are utilised by bats for roosting, hibernation and maternity sites (Brown et al., 1997). Gibbons and Lindenmayer (2002) estimate that 43 of the 65 species of microbats that occur in Australia utilise tree hollows. Bats can occupy small hollows when roosting singly, but may require large hollows for communal roosts, in which more than, 200 individuals can occupy a single large tree (Tidemann and Flavel, 1987). Lunney et al., (1988) found that Gould’s Long-eared Bat (Nyctophilus gouldii) roosted only in trees with a dbh greater than 80 cm and that roost sites were changed daily, so that multiple hollow-bearing trees were required for each individual or colony. Taylor and Savva (1988) found that trees with a dbh greater than 120 cm were preferred as roost sites.
Despite the importance of large hollow-bearing trees to many bat species, stand level attributes such as the number of hollow-bearing trees or tree dbh will not necessarily be correlated with the presence or abundance of bats. This is because bats have large home ranges and may travel two or more kilometres from roost to foraging site (Brown et al., 1997, Smith et al., 1994). The mobility and communal organisation of many bat species also means that unlike arboreal marsupials, bats will tolerate the clustering of hollow-bearing trees used for roosting (Smith et al., 1994). The abundance of hollow dependent bats has been positively correlated with the maximum dbh of unburnt trees (Andrews et al., 1994), although this attribute was more directly a measure of understorey complexity than hollow development.

In addition to hollows within trees, bats may also roost in crevices under peeling bark. Dead trees are an important source of this type of roost site, forming a series of crevices as the bark separates from the main stem. These crevices are preferred roosting sites for a number of bat species (Taylor and Savva, 1988; Lunney et al., 1988). Lunney et al., (1988) also found that the distribution of size classes amongst overstorey species was a significant factor controlling roost selection. In their study, Gould’s long-eared bat preferentially roosted in large trees of particular overstorey species, a choice which in part reflected the production of decorticating bark, and in part the development of hollows.

**WATER**

Easy access to water is important for bats. Species that hibernate in cool, high elevation forests need to arise periodically in winter because body water turnover is higher than energy turnover. Water sources close to hibernating sites ensure that stored fat is not wasted in long flights (Smith et al., 1994). Similarly in semi-arid zones most bat species are restricted to moist environments associated with rivers, lakes and dams (Lumsden and Bennett, 1995). Although roost sites have been shown to be associated with proximity to water (Tidemann and Flavel, 1987) no study has correlated bat abundance or richness with distance to permanent water source.
4.3.5 Habitat resources used by reptiles

Overview

In contrast to the detailed documentation of the habitat requirements of birds and mammals in south-east and south-west Australia relatively fewer studies have related reptile abundance and richness to particular habitat components (Woinarski et al., 1997, Brown and Nelson, 1993). From available literature the following habitat components have been associated with reptile abundance:

- **Basking sites**: for body temperature regulation;
- **Logs and woody debris**: for shelter, basking and foraging for prey;
- **Standing dead trees**: for shelter and foraging;
- **Litter**: for foraging and cover;
- **Shrubs**: for cover, invertebrate prey, and foliage input to the litter layer.

The structural attributes used to quantify these habitat components are listed in Table 8. The individual habitat components are then discussed below.

Table 8: Structural attributes significantly (p<0.05), and positively associated with the presence, abundance or richness of one or more species of reptiles in eucalypt forests (F) or woodlands (W). (-) indicates a study which reported a negative association for some species.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abundance of large dead trees</td>
<td>(F) Andrews et al., 1994</td>
</tr>
<tr>
<td>Mean dbh of dead trees</td>
<td>(F) Brown and Nelson, 1993 (-)</td>
</tr>
<tr>
<td>% cover or other measure of shrub abundance</td>
<td>(F)(W) Brown, 2001, Bauer et al., 2000, Hadden and Westbrooke, 1996, Arnold et al., 1987</td>
</tr>
<tr>
<td>% cover or other measure of log abundance</td>
<td>(F) Bauer et al., 2000, Andrews et al., 1994, Brown and Nelson, 1993(-)</td>
</tr>
<tr>
<td>% cover of grass</td>
<td>(F) Brown, 2001, Brown and Nelson, 1993 (-)</td>
</tr>
<tr>
<td>Presence of a cryptogam crust</td>
<td>(W) Smith et al., 1996</td>
</tr>
<tr>
<td>% cover of bare ground</td>
<td>(F) Brown and Nelson, 1993 (-)</td>
</tr>
<tr>
<td>% rock cover</td>
<td>(F) Fanning, 1995</td>
</tr>
</tbody>
</table>
Reptile habitat components

BASKING SITES
The availability of insolation at suitable basking sites is an important component of reptile habitat because reptiles depend on external heat sources to regulate their body temperature (Bauer et al., 2000; Andrews et al., 1994; Brown and Nelson, 1993). To absorb heat, many reptile species utilise rock outcrops for basking sites, and Fanning (1995) found that reptile species richness was significantly higher at sites containing these ground features. Lunney et al., (1991) demonstrated the role of canopy structure in regulating insolation. They found that the uniformity of tree size and shape in regrowth forests in south-east NSW reduced the amount of sunlight reaching the ground. The rarity of normally common lizard species in these regrowth forests was attributed to a shortage of basking sites.

Brown and Nelson (1993) found that the insolation requirements of reptile species can differ significantly, indicating that a diversity of reptiles will require a range of lighting conditions from direct to semi-shaded (Woinarski et al., 1997). These conditions are most likely to be provided by a patchy canopy (Lunney et al., 1991). Patchiness can be quantified by attributes such as the coefficient of variation of canopy cover or the coefficient of variation of tree dbh. To date no studies have attempted to correlate these attributes with reptile abundance or richness.

LOGS AND WOODY DEBRIS
Logs and woody debris provide foraging, basking, nesting and hibernation sites for reptiles (Brown and Nelson, 1993; Slip and Shine, 1988; Web, 1985). Large logs, which hold moisture, can also act as refuges for reptiles during drought and fire (Andrews et al., 1994). Reptile abundance and richness have been significantly correlated with attributes that quantify log abundance such as percentage cover of logs, log length and number of logs (Bauer et al., 2000; Andrews et al., 1994). However, in the Mountain Ash forests of the Victorian Highlands, Brown and Nelson (1993) found a negative relationship between the number of logs and reptile abundance. They attributed this to the large number of logs being indicative of older forests, with a more developed understorey,
which provides little insolation for reptiles.

**STANDING DEAD TREES**

Hollows and crevasses in the wood and bark of standing dead trees can provide important shelter and foraging sites for arboreal reptile fauna (Bauer et al., 2000). Reptile richness was correlated with the numbers of dead trees (Andrews et al., 1994), although in Mountain Ash forest, Brown and Nelson (1993) found reptile abundance was negatively correlated with the mean diameter of dead trees.

**LITTER**

Litter provides ground foraging reptiles with invertebrates for food, and a range of sites for shelter and basking. Decorticating bark is an important component of litter for ground dwelling reptiles. Brown and Nelson (1993) found lizards preferentially foraged in bark compared with other litter components. However, no studies have attempted to relate the abundance of reptiles with measures of decorticating bark. Larger predatory reptiles such as varanids are indirectly dependent on the litter layer because they require a source of ground reptiles for prey (Recher and Lim, 1990).

Attributes that measure the quantity of litter such as percentage cover and litter depth have been significantly correlated with reptile abundance (Brown, 2001; Smith et al., 1996; Brown and Nelson, 1993). However, in Mountain Ash forests a sparse litter layer can be indicative of reduced canopy and / or shrub cover, which can increase insolation at ground level. In these wet environments, the abundance of some heliothermic reptiles, has been negatively correlated with litter depth (Brown and Nelson, 1993).

**SHRUBS**

Reptiles can be significantly more abundant in forests with well-developed shrub and ground vegetation layers (Brown, 2001). Shrubs provide invertebrate prey for reptile species (Woinarski et al., 1997), and contribute plant material to the litter layer. A dense shrubby understorey produces a thick moist litter layer important for some litter dwelling skinks (Woinarski et al., 1997); whereas a simplified shrub layer produces a sparse litter layer, and in white cypress pine
(Callitris glaucophylla) forests this can reduce reptile abundance and richness (Bauer et al., 2000).

Attributes such as shrub richness, percentage cover of shrubs, and numbers of shrub stems have been used to quantify shrub resources for reptiles (Brown, 2001; Bauer et al., 2000; Hadden and Westbrooke, 1996; Arnold et al., 1987)

### 4.3.6 Habitat resources used by amphibians

**Amphibian habitat components**

The ecological requirements of Australian amphibians are poorly documented compared with other faunal groups (Hazell et al., 2001; Lemckert and Brassil, 2000). Despite this lack of information three habitat components have been identified as important for the survival of amphibians:

- **Proximity to free water**: to maintain body moisture and for reproduction;
- **Vegetation cover**: to maintain a moist micro-environment and for shelter;
- **Ground debris**: for shelter and refuge sites.

Of these three habitat components water is the most important (Bauer et al., 2000; Parris and McCarthy, 1999; Fanning, 1995; Smith et al., 1994). This is because amphibians need to maintain a moist skin, and usually require free water for reproduction and the subsequent development of their young (Smith et al., 1994); although some species can breed in moist litter or boggy seepages (Andrews et al., 1994).

Overstorey canopy cover and understorey vegetation are important to amphibians through their role in maintaining a moist micro-environment (Parris, 2002). These features provide cover, which ameliorates the light, temperature, moisture and humidity conditions at ground level (Ferraro and Burgin, 1993). Understorey vegetation and ground debris such as logs, rocks and litter also provide shelter and refuge from predation (Ferraro and Burgin, 1993). The presence of these features at the aquatic terrestrial boundary can be critical during metamorphosis when the process of tail resorption limits mobility. Hazell et al., (2001) found that frog species richness was significantly higher when
grass tussocks were present in the riparian zone, and was negatively correlated with the % cover of bare ground. Similarly, Bauer et al., (2000) attributed the scarcity of amphibians in the white cypress pine forests of south-west NSW to a lack of aquatic resources and of suitable cover adjacent to the few water bodies that were present.

Relatively few studies have correlated the presence or abundance of amphibian species with vegetation structural attributes. This may in part reflect the scarcity of amphibians in many forest and woodland systems, but also the reliance of surveys on opportunistic sightings, which limits any statistical analysis of habitat associations (Andrews et al., 1994; Smith et al., 1994). Those structural attributes used to quantify amphibian habitat components are listed in Table 9.

Table 9: Structural attributes significantly (p<0.05), and positively associated with the abundance or richness of species of amphibians in eucalypt forests (F), and woodlands (W). (−) indicates a study which reported a negative association.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>% cover of litter</td>
<td>(F) Smith et al., 1994</td>
</tr>
<tr>
<td>Presence of tussocks in the riparian zone</td>
<td>(F)(W) Hazell et al., 2001</td>
</tr>
<tr>
<td>Stream size</td>
<td>(F) Parris and McCarthy, 1999</td>
</tr>
<tr>
<td>% cover of bare ground in riparian zone</td>
<td>(F)(W) Hazell et al., 2001 (−)</td>
</tr>
<tr>
<td>% of water body containing emergent vegetation</td>
<td>(F)(W) Hazell et al., 2001</td>
</tr>
<tr>
<td>Log cover (classes)</td>
<td>(F) Andrews et al., 1994</td>
</tr>
</tbody>
</table>

4.3.7 Habitat resources used by invertebrates

Overview
The above ground component of eucalypt forests and woodlands provide three broad habitats for invertebrates - overstorey trees, understorey shrubs and grasses, and the ground layer of litter, woody debris and bare earth (Majer et al., 1997). Within these habitats the spatial scale at which invertebrate diversity occurs can be small, and individual plants or forest patches may support distinct invertebrate communities reflecting differences in bark, foliage and litter characteristics (Evans et al., 2003; Doherty et al., 2000; Majer et al., 1997; Recher et al., 1996). At the single tree scale, the richness of invertebrate
species is primarily determined by the structural complexity of the plant, its biochemical defences against attack and its foliage nutrient levels (Majer et al., 1997). The richest invertebrate communities have been associated with large old eucalypts because these are structurally more complex than younger trees (Recher et al., 1996). At a stand scale invertebrate diversity may reflect floristic diversity because different invertebrates are adapted to feeding on different plant species (Recher et al., 1996). Plant phenology also influences the abundance and diversity of invertebrates, with many invertebrate taxa responding to periods of new leaf production and flowering (Majer et al., 1997).

Four habitat components have been used to characterise the resources used by invertebrates:

- **Overstorey foliage and flowers;** a source of edible leaf material, leaf sap, nectar and pollen and shelter sites within foliage;
- **Bark;** for shelter and food resources such as litter, exudates and prey;
- **Shrubs and ground vegetation;** for foliage and flower resources, and as attachment points for web building spiders.
- **Litter and woody debris;** for food and prey and the provision of shelter and a suitable micro-climate

The removal or simplification of some, or all, of these habitat components can reduce invertebrate richness and abundance (Bauer et al., 2000). However, some invertebrates, such as ants, cockroaches and beetles can respond positively to habitat simplification (Abensperg-Traun, 1996), such as the creation of canopy and understorey openings that increase ground insolation levels (Oliver et al., 2000; Andersen, 1986). Invertebrate diversity is therefore likely to be greatest in heterogeneous habitats containing a variety of structural arrangements, ranging from open areas beneath overstorey trees to dense thickets of tall shrubs.

The structural attributes used to quantify invertebrate habitat components are listed in Table 10. The individual habitat components are discussed below.
Table 10: Structural attributes significantly (p<0.05), and positively associated with the presence, abundance or richness of one or more orders or species of invertebrates in eucalypt forests (F) or woodlands (W). (-) indicates a study which reported a negative association for some orders.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>% cover of overstorey</td>
<td>(F) York, 2000</td>
</tr>
<tr>
<td>Number of overstorey stems</td>
<td>(F)(W) Oliver et al., 2000, Bromham et al., 1999</td>
</tr>
<tr>
<td>Eucalypt subgenera</td>
<td>(F)(W) Woinarski and Cullen, 1984</td>
</tr>
<tr>
<td>% cover of subcanopy</td>
<td>(F) Oliver et al., 2000</td>
</tr>
<tr>
<td>% cover or abundance of shrubs</td>
<td>(F) York, 2000, Bromham et al., 1999, Apensberg-Traun et al., 1996</td>
</tr>
<tr>
<td>Measure of litter abundance</td>
<td>(F)(W) York, 2000&lt;sup&gt;A&lt;/sup&gt;, Oliver et al., 2000&lt;sup&gt;B&lt;/sup&gt;, Bromham et al., 1999&lt;sup&gt;AB&lt;/sup&gt;</td>
</tr>
<tr>
<td>Presence of a cryptogam crust</td>
<td>(W) Apensberg-Traun et al., 1996 (-)</td>
</tr>
<tr>
<td>% cover bare ground</td>
<td>(F)(W) Oliver et al., 2000, Bromham et al., 1999 (-)</td>
</tr>
</tbody>
</table>

**Invertebrate habitat components**

**OVERSTOREY FOLIAGE AND FLOWERS**

The foliage and flowers of eucalypt forests and woodlands support some of the richest invertebrate communities in the world (Majer et al., 1997). Recher et al., (1996) recorded 976 species of canopy invertebrates on two eucalypt tree species in south-east Australia, and 683 species on two tree species from south-west Australia. Increasing diversity and abundance of invertebrates has been correlated with increasing foliar nutrient concentrations (Recher et al., 1996). At the level of eucalypt subgenera this results in *Symphyomyrtus* species having significantly richer invertebrate assemblages than *Monocalyptus* species. Woinarski and Cullen (1984) found that *Symphyomyrtus* species had six times the densities of lerp forming psyllids, and two times the density of total invertebrates when compared with *Monocalyptus* species.

Significant differences in arthropod composition may also occur between different eucalypt species or between trees of the same species (Majer et al., 1997). This reflects differences in the type and quantity of foliage produced (Recher et al., 1996), in the timing of flowering and leaf production (Majer et al., 1997, Woinarski and Cullen, 1984), and in the association of some
invertebrates with particular plant species. For example, the larvae of some butterflies feed on the leaves of particular mistletoe species, which in turn parasitise particular eucalypt species (Woodgate et al., 1994). Overstorey species richness, the distribution of basal area amongst overstorey species and tree dbh are attributes likely to describe foliage and flower resources for invertebrates. However, to date no study has correlated these attributes with invertebrate abundance or richness.

**SHRUBS AND GROUND VEGETATION**

Shrubs and ground vegetation provide foliage and flower resources, which may be critical for the juvenile or larval stage of many invertebrates (Candusio, 1996). Shrubs also provide attachment points for web-building spiders (Candusio, 1996, Uetz, 1990). Greenstone (1984) found that as the diversity of the shrub layer increased, so did the potential number of attachment points for webs, thereby increasing the diversity of web building spiders. Attributes such as percentage cover of shrubs, and number of shrubs, are direct measures of the shrub resource and have been significantly related to invertebrate diversity (York, 2000, Bromham et al., 1999). Lichen cover and percentage cover of bare earth are indicative of the development of ground vegetation and have also been significantly related to invertebrate abundance (Oliver et al., 2000, Bromham et al., 1999, Apensberg-Traun et al., 1996).

**BARK**

The bark of eucalypt forest and woodland trees supports a rich invertebrate fauna, with up to 300 species of invertebrates having been associated with the bark of a single eucalypt tree (Recher et al., 1996). The abundance and richness of bark invertebrate communities differs between tree species reflecting differences in bark type and tree phenology (Dickman, 1991). In gum-barked eucalypt species invertebrate abundance fluctuates seasonally in response to bark shedding, whereas more stable populations are maintained year round on rough barked species (Dickman, 1991). Within the same tree species the richest invertebrate communities have been associated with large trees reflecting their complex bark structure and large surface area of bark (Recher et al., 1996). Attributes likely to characterise bark resources are the distribution of basal area amongst overstorey species, and mean dbh. However,
no studies appear to have correlated these, or any other measures of bark resources with invertebrate abundance or diversity.

**LITTER**

The constituents of litter - leaves, branches, logs and bark - provide food and shelter resources for a distinct invertebrate fauna (Majer *et al.*, 1997). This ground fauna includes collemboila and acarina species which feed on decomposing plant material (Majer *et al.*, 1997, Candusio, 1996), and which have been shown to increase in abundance with development of the litter layer and increasing soil organic matter (Majer *et al.*, 1997). These detrivores may in turn support a diversity of predator invertebrates including centipedes and spiders for which litter complexity is also critical. Several studies in deciduous temperate forests have correlated spider species richness with litter depth and complexity, indicating that spider habitat increases as the abundance and diversity of spaces within the litter increases (Uetz, 1990).

Complexity of litter reflects the diversity of plants contributing material to the litter layer. Andersen (1986) found that the complex and dense litter produced by woodland vegetation supported significantly more individuals and species of ant than the more uniform litter produced in adjacent heath. Similarly Martin and Major (2001) found that litter components in woodland were more diverse than in pasture and that this in part explained the increased abundance of wolf spiders in the woodland. Attributes such as the number of overstorey trees, the number of shrubs and the percentage cover of ground vegetation will be indirect measures of litter abundance and complexity and have in turn been associated with the diversity of ground invertebrates (Bromham *et al.*, 1999).

A reduction in the abundance and richness of overstorey or understorey plants is likely to reduce the abundance of litter and the variety of food and habitat resources that it provides (Andersen, 1986). Loss of plant and litter cover may also leave the ground less sheltered reducing the availability of ground moisture. These conditions can lead to significant losses in invertebrate abundance and richness (Bromham *et al.*, 1999, Holt *et al.*, 1988). However, for some invertebrate communities such as ants, overall species richness may be
maintained by the addition of new species not present on sites with a well
developed litter layer (York, 2000). These invertebrate species respond to
increased ground insolation and the presence of patches of bare earth.

**4.4. Synthesis: key structural attributes**

**4.4.1 Generic attributes**

In this section the various structural attributes identified for each of the different
faunal groups in section 4.3, are combined to produce a set of key attributes.
This set of attributes provides a basis for characterising the habitat
requirements of all faunal groups, and hence also for characterising the
potential of a particular stand of vegetation to support a diversity of fauna. To
produce the key set, similar attributes have been grouped, and then
represented by a single generic attribute. For example the attributes,
“percentage cover of litter layer”, “litter depth”, “litter biomass”, and “litter
volume” were all combined under the generic attribute “percentage cover or
other measure of litter abundance”. In this example “litter abundance” is the key
attribute, while the fact that different studies have quantified it in a number of
ways is of secondary importance. Similarly the various indices of decorticating
bark used by different studies are combined under the generic attribute
“abundance of decorticating bark”.

The key set of attributes is presented in Table 11. The shaded boxes in this
table indicate which faunal groups were significantly associated with each
attribute. This information could provide a basis for ranking attributes in terms of
their value as indicators of faunal diversity, with attribute value increasing with
the number of faunal groups in which an association was observed. However,
this approach was not adopted for two reasons:

a) Not all faunal groups have been studied with the same intensity, so that the
occurrence of an attribute across only a few faunal groups may indicate a lack
of suitable studies, rather than an attribute that is rare or relatively unimportant.
b) An attribute may define part of a niche utilised by members of a given faunal
group, so that the presence of a number of rare attributes may be critical to
sustaining a diversity of faunal species.
Table 11: Attributes significantly associated with the presence or abundance of major faunal groupings in temperate eucalypt forests (F) or woodlands (W)

<table>
<thead>
<tr>
<th>Stratum</th>
<th>Attribute</th>
<th>Birds</th>
<th>Arboreal marsupials</th>
<th>Ground mammals</th>
<th>Bats</th>
<th>Reptiles</th>
<th>Amphibians</th>
<th>Invertebrates</th>
</tr>
</thead>
<tbody>
<tr>
<td>Overstorey</td>
<td>Number of overstorey stems (F)(W)</td>
<td>S</td>
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<tr>
<td></td>
<td>Number or basal area of large trees (F)</td>
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<td></td>
<td>Overstorey dbh (F)</td>
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<td></td>
<td>Basal area of overstorey stems (F)(W)</td>
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<tr>
<td></td>
<td>Overstorey diversity or species richness (F)</td>
<td>S, Fn, C</td>
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<tr>
<td></td>
<td>Eucalypt species richness (F)</td>
<td>Fn, C</td>
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<td></td>
<td>Species distribution among eucalypt subgenera (F)(W)</td>
<td>Fn, C</td>
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<tr>
<td></td>
<td>Abundance of floral resources (F)</td>
<td>Fn, C</td>
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<td></td>
<td>Mistletoe density (W)</td>
<td>S, Fn, C</td>
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<tr>
<td></td>
<td>Abundance of decorticating bark (F)</td>
<td>S, Fn, C</td>
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<td></td>
<td>Measures of foliar nutrients (F)(W)</td>
<td>Fn</td>
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<td></td>
<td>Dbh distribution amongst overstorey species (F)</td>
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<td></td>
<td>% cover canopy trees (F)(W)</td>
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<td></td>
<td>Canopy height (F)(W)</td>
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<td></td>
<td>Canopy surface area and volume (F)(W)</td>
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<tr>
<td></td>
<td>Number of hollow-bearing trees (F)</td>
<td>S, Fn</td>
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<td></td>
<td>Abundance of large dead trees (F)</td>
<td>S, Fn</td>
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<td></td>
<td>Mean dbh of dead trees (F)</td>
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<tr>
<td></td>
<td>Spatial distribution of hollow-bearing trees (F)</td>
<td>S</td>
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<tr>
<td>Mid-storey</td>
<td>Mid-storey height (F)</td>
<td>S</td>
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<td></td>
<td>Canopy – mid-storey height difference (F)</td>
<td>S</td>
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<td></td>
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<tr>
<td></td>
<td>% cover of mid-storey (F)</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Shrub layer</td>
<td>Shrub species richness (F)(W)</td>
<td>Fn, C</td>
<td></td>
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<tr>
<td></td>
<td>% cover shrubs or other abundance measure (F)(W)</td>
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<tr>
<td></td>
<td>Vertical density of shrub layer foliage (F)</td>
<td>S</td>
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</tr>
<tr>
<td>Herb layer</td>
<td>% cover forbs and grasses (F)(W)</td>
<td>S</td>
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<td></td>
</tr>
<tr>
<td>Ground layer</td>
<td>% cover or other measure of litter abundance (F)(W)</td>
<td>S, Fn</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>% cover or other measure of log abundance (W)</td>
<td>S, Fn</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>% cover bare ground (F)(W)</td>
<td>S</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>% rock cover (F)</td>
<td>S</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>% cover of cryptogams (W)</td>
<td>S, C</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>% cover of debris (litter, logs and rocks) (F)</td>
<td>S</td>
<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
</tbody>
</table>
All attributes forming the key set were therefore accepted as potentially important for characterising habitat in Australian temperate forests and woodlands. However, it should be possible to reduce this key set on the basis of predicted or established correlations between the attributes, an issue that is addressed in chapter 6 of this thesis.

Table 11 also indicates that the 34 attributes identified in this review describe elements of ecosystem structure (the spatial arrangement of components), composition (the identity and variety of components) and function (the types and rates of ecological processes). However, it is unlikely that these 34 attributes will capture all elements of ecosystem structure, function and composition because these attributes were initially identified on the basis of their role in describing fauna habitat. For example a key functional attribute such as the presence of adequate regeneration was not identified from the review process. The 34 attributes must therefore be seen as a significant but not necessarily definitive set of attributes with which to characterise Australian temperate ecosystems.

To address this situation the 34 attributes were considered in association with additional attributes identified in the review presented in chapter 2 of this thesis (see Table 1). Taken together these attributes constitute a comprehensive set with a demonstrated association with biodiversity. This set provided the basis for developing an operational set of stand level attributes, which could be used to collect data in the field. This operational set is summarised in Table 12. Some attributes in the comprehensive set were excluded from the operational set on the grounds they were strongly seasonal (e.g. the abundance of floral resources, the foliage cover and richness of herbaceous species), or time consuming to quantify (e.g. foliage height diversity, measures of foliar nutrients, spatial distribution of trees, amount of decorticating bark), and were therefore inappropriate for consideration as part of a rapid assessment index.

In the next chapter I describe how the operational set of attributes was used to collect data from a representative set of stands within a 15,000 km$^2$ study area.
Table 12: An operational set of stand level attributes, for the collection of field data. Attributes were based on a comprehensive set identified by reviewing the literature concerning forest and woodland structure (Chapter 2), and fauna–habitat relationships in Australian temperate ecosystems (this Chapter).

<table>
<thead>
<tr>
<th>Stratum</th>
<th>Structural element</th>
<th>Structural attribute</th>
</tr>
</thead>
<tbody>
<tr>
<td>Overstorey</td>
<td>Canopy (&gt; 6m) foliage cover</td>
<td>Canopy projective foliage cover (%), canopy projective foliage cover coefficient of variation,</td>
</tr>
<tr>
<td></td>
<td>Stand height</td>
<td>Mean height of three tallest trees</td>
</tr>
<tr>
<td></td>
<td>Stand density</td>
<td>Tree basal area, number of overstorey stems</td>
</tr>
<tr>
<td></td>
<td>Overstorey dbh</td>
<td>dbh, quadratic mean dbh, standard deviation of dbh, number of stems &gt; threshold dbh (large trees)</td>
</tr>
<tr>
<td></td>
<td>Overstorey species</td>
<td>Indigenous tree species richness, distribution of basal area amongst indigenous tree species</td>
</tr>
<tr>
<td></td>
<td>Mistletoe</td>
<td>Presence/absence of mistletoe, basal area of trees with mistletoe</td>
</tr>
<tr>
<td></td>
<td>Dead trees</td>
<td>Basal area of dead trees, number of dead trees, mean dbh of dead trees</td>
</tr>
<tr>
<td></td>
<td>Hollow trees</td>
<td>Number of hollow bearing trees, number of trees with small, medium and/or large hollows, tree hollow size diversity</td>
</tr>
<tr>
<td>Understorey</td>
<td>Lifeforms</td>
<td>Indigenous lifeform richness, projective foliage cover (%) within lifeforms, indigenous lifeform diversity</td>
</tr>
<tr>
<td></td>
<td>Perennial species</td>
<td>Indigenous perennial species richness, indigenous shrub species richness</td>
</tr>
<tr>
<td></td>
<td>Understorey foliage cover</td>
<td>Projective foliage cover of indigenous and exotic vegetation (%) in 4 height classes; &lt; 0.5m, 0.5m-2m, 2m-4m, 4m-6m</td>
</tr>
<tr>
<td></td>
<td>Overstorey regeneration</td>
<td>Number of regenerating stems, distribution of regeneration amongst overstorey species, projective foliage cover (%) of regeneration &lt; 2m, projective foliage cover of regeneration (%) 2-6m</td>
</tr>
<tr>
<td>Ground layer</td>
<td>Litter</td>
<td>Litter cover (%), litter biomass tha⁻¹</td>
</tr>
<tr>
<td></td>
<td>Ground cover</td>
<td>Cover of bare earth (%), cover of rocks (%), cover of cryptogams (%)</td>
</tr>
<tr>
<td></td>
<td>Coarse woody debris</td>
<td>Log cover (%), total log length, log length by 3 decay classes, total log volume, log volume by 3 decay classes, length of logs &gt; threshold diameter (large logs), volume of logs &gt; threshold diameter (large logs)</td>
</tr>
</tbody>
</table>
Chapter 5: A method for collecting structural attribute data in a representative sample of stands

Chapter summary
This chapter describes the method used to collect data for quantifying a comprehensive set of structural attributes in 144 plots, at 48 study sites located within a 15,000 km² study area in the South-eastern Highlands Bioregion of Australia. To identify a representative set of sites, the study area was stratified by the factors considered most likely to contribute to variation at the landscape scale. This produced 24 strata on the basis of three vegetation communities, two catchments, two levels of rainfall and two levels of condition. Two sites (replicates) were measured within each of these 24 strata. All sites were clearly defined stands located within an agricultural matrix, or within a larger tract of native vegetation. At the site scale, three plots were systematically established, from a random starting point, to provide an unbiased sample of the target stand. At the plot level, measurements incorporated three different sampling methods - 50mx20m quadrat, 20mx20m quadrat and 50m transect - so that a comparison could be made between the precision and efficiency of the different methods.

5.1 Introduction
In this chapter I address stages two and three of the methodology for developing an index of structural complexity, which was outlined in Chapter 3. To do this, I describe a method for collecting data based on the comprehensive set of structural attributes identified in the previous chapter, and the sampling approach used to apply this method to a representative set of study sites. The purpose of collecting these data was to identify which were the most important, or core, attributes in the comprehensive set. This was done by identifying those attributes that best distinguished between study sites, and were most efficient to measure in the field, and by reducing the number of attributes on the basis of correlations between attributes. This analysis is presented in the next chapter.
5.2 Study area

The study was located within the woodlands and dry sclerophyll forests of south-eastern Australia. These communities occur largely on private or leasehold land and have been extensively cleared or modified in the past. They now exist as remnants and patches of regrowth, in a fragmented landscape dominated by agriculture (National Forest Inventory, 1998). A metric for ranking stands in terms of their condition and potential contribution to biodiversity, such as a structural complexity index, is particularly relevant to these communities. If properly designed, it could provide an objective basis for assessing applications to clear native vegetation (e.g. Gibbons et al., 2004), and for allocating financial incentives to improve or maintain the condition of remnants on private land (e.g. Parkes et al., 2003). It could also form the basis for developing a practical tool for guiding and improving management within the context of private ownership (e.g. Goldney and Wakefield, 1997).

The study area was located within the South-eastern Highlands Bioregion of Australia (Thackway and Creswell, 1995), and is shown in Figure 3. The area covered 15,000 km\(^2\), and ranged in latitude from -35\(^\circ\) 38’ 46” to -34\(^\circ\) 17’ 8”, and in longitude from 148\(^\circ\) 35’ 28” to 149\(^\circ\) 45’ 6”. It ranged in altitude from 500m to 1100m, and included sections of the Murrumbidgee and Lachlan river catchments.

The woodlands and dry sclerophyll forests of south-eastern Australia occur in the relatively dry environments of the tablelands and western slopes of New South Wales where annual rainfall is between 400-800mm. The vegetation typically comprises a single canopy layer dominated by *Eucalyptus* and *Corymbia* species, over an understorey of forbs, perennial grasses and sclerophyllous shrubs (McIntyre et al., 2002). In woodlands, overstorey trees are often more widely spaced than in dry sclerophyll forests, so that woodland projective foliage cover is usually less than 30% whereas in dry sclerophyll forest it ranges from 30-70% (Specht, 1970).
Figure 3: The study area covered 15,000 km$^2$, and ranged in latitude from -35° 38' 46" to -34° 17' 8", and in longitude from 148° 35' 28" to 149° 45' 6". It ranged in altitude from 500m to 1100m, and included sections of the Murrumbidgee and Lachlan river catchments. A total of 144 plots in 48 sites were used to sample variation across the study area.
Three communities, one woodland and two dry sclerophyll forest, were chosen as the focus for the study (Table 13). These communities were defined in terms of their dominant overstorey species, and were selected because they often intergrade to form a topographic sequence from ridge top to lower slope and valley floor (Costin, 1954).

Table 13: The three vegetation communities selected for this study. Communities were defined in terms of their dominant overstorey species, and were chosen because they intergrade to form a topographic sequence from ridge top to lower slope and valley floor. These communities incorporate one or more of the ecosystem types classified by Thomas et al., (2000)\textsuperscript{A}, and now exist as small remnants and patches of regrowth in a fragmented landscape dominated by agriculture\textsuperscript{B}.

<table>
<thead>
<tr>
<th>Vegetation community</th>
<th>Overstorey species</th>
<th>Topographic position</th>
<th>Ecosystem types\textsuperscript{A}</th>
<th>% pre –1750 area cleared\textsuperscript{B}</th>
</tr>
</thead>
<tbody>
<tr>
<td>Yellow Box-Red Gum</td>
<td>\textit{E. melliodora} \textit{E. blakely} \textit{E. bridgesiana}</td>
<td>Lower slopes and valley floors</td>
<td>CRA 161</td>
<td>97.8%</td>
</tr>
<tr>
<td>grassy woodland</td>
<td></td>
<td></td>
<td>CRA 160</td>
<td>96.1%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>CRA 159</td>
<td>96.6%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>CRA 154</td>
<td>95.2%</td>
</tr>
<tr>
<td>Broadleaved Peppermint-Brittle Gum</td>
<td>\textit{E. dives} \textit{E. mannifera} \textit{E. macrocryncha} \textit{E. dalrympleana}</td>
<td>Middle and lower slopes</td>
<td>CRA 111</td>
<td>89.6%</td>
</tr>
<tr>
<td>dry sclerophyll forest</td>
<td></td>
<td></td>
<td>CRA 109</td>
<td>54.8%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>CRA 108</td>
<td>40.0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>CRA 80</td>
<td>47.5%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>CRA 81</td>
<td>37.0%</td>
</tr>
<tr>
<td>Scribbly Gum-Red Stringybark</td>
<td>\textit{E. rossii} \textit{E. macrorhyncha}</td>
<td>Upper slopes and ridges</td>
<td>CRA 114</td>
<td>52.8%</td>
</tr>
<tr>
<td>dry sclerophyll forest</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

5.3 Data collection

5.2.1 Sampling design

Identifying study sites

To collect a representative data set from the 15,000 km\textsuperscript{2} study area, a sampling design was employed which balanced the allocation of resources between effort spent in sampling between site variation, and effort spent in sampling within site variation. To sample between site variation, the study area was stratified by the following 4 variables, which were considered most likely to contribute to
variation at the landscape scale:

1. Vegetation community; 3 groups based on overstorey species composition.
   a) Scribbly Gum - Red Stringybark community b) Broadleaved Peppermint - Brittle Gum community c) Yellow Box - Red Gum community.

2. Catchment; 2 groups based on location within either the Lachlan river catchment or the Murrumbidgee river catchment. A shortage of suitable sites in the Lachlan catchment meant that 6 study sites were located in the adjoining Shoalhaven and Wollondilly catchments.

3. Rainfall; 2 groups based on whether mean annual rainfall was high (700-800mm/yr), or low (600-700mm/yr). Rainfall was predicted using ESOCLIM (Hutchinson, 1989) and a 250m Digital Elevation Model.

4. Condition; 2 groups indicating the degree of modification since European settlement. Sites were classified as either unmodified or modified. Unmodified sites were ones in which the effects of any modification since European settlement appeared negligible. These sites contained little evidence of past clearing or logging in the form of stumps, and were typically dominated by senescing trees although a regrowth component could also be present. Modified sites were either regrowth stands on land formerly cleared for agriculture, or sites in which a large proportion of the original overstorey had been removed as evidenced by the presence of stumps, parts of felled trees, and a much reduced canopy cover.

For the Yellow Box-Red Gum community there were no unmodified stands. For this community “unmodified” indicated stands whose structure and composition were relatively less modified than other stands.

These 4 variables produced 3x2x2x2 = 24 strata. Within each stratum two study sites (replicates) were subjectively identified with the aid of a New South Wales National Parks and Wildlife Service database of high quality sites (Rehwinkle, 2002), a recently completed forest classification and mapping GIS layer (Thomas et al., 2000), considerable local knowledge and expert advice from
Nick Webb (Environment ACT) and Jacqui Stol (CSIRO Sustainable Ecosystems), and my own field reconnaissance observations. In total 48 sites were used to sample variation across the study area (Figure 3 and Table 14). Sites existed either as an isolated stand within an agricultural matrix, or as a clearly defined stand within a larger tract of native vegetation.

Patch size was not included in the stratification design, because reconnaissance surveys indicated that this was not a good predictor of site condition, an observation supported by Prober and Thiele (1995). For example, modified stands could occur as large or small regrowth patches on land which had been cleared for pasture and then abandoned, and unmodified stands could occur as small isolated remnants apparently buffered against invasion by exotic vegetation by a relatively infertile environment, or as large intact patches protected within a reserve system.

Establishing plots at each site

At the site scale, data for a range of structural attributes were obtained by establishing three measurement plots within the target stand at each site. A pilot study indicated that three plots was an efficient number for estimating means and variances for a range of stand variables. To capture within stand variation, plots were located on a stand transect running diagonally across the direction of the main environmental gradient (assumed to be topography), and passing through the centre of the stand. To provide unbiased estimates of stand variables, plots were evenly distributed along the stand transect starting from a random point 1-100m from the stand edge (Figure 4).
Table 14: 48 study sites across 24 strata. Strata were identified on the basis of vegetation community (YR=Yellow Box–Red Gum; PB=Broadleaved Peppermint–Brittle Gum, SR=Scribbly Gum–Red Stringybark), catchment (L=Lachlan, M=Murrumbidgee). For logistical reasons 2 sites were located in the Shoalhaven (S) catchment, and 4 sites in the Wollondilly (W) catchment.), rainfall (H=700-800mm/yr, L=600-700mm/yr), and condition (U=relatively unmodified, M=modified). (TSR=Travelling Stock Reserve).

<table>
<thead>
<tr>
<th>Site name</th>
<th>Location</th>
<th>Community</th>
<th>Catchment</th>
<th>Rainfall</th>
<th>Condition</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Easting</td>
<td>Northing</td>
<td>YR, PB, RM</td>
<td>L / M / S / W</td>
<td>H / L</td>
</tr>
<tr>
<td>“Braeside”, via Tharwa, ACT</td>
<td>685465</td>
<td>6070030</td>
<td>YR</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>“Kama” leasehold, William Howell Drive, ACT</td>
<td>683759</td>
<td>6095430</td>
<td>YR</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>Mullion Creek TSR #60, Mountain Creek Rd.</td>
<td>670926</td>
<td>6109703</td>
<td>YR</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>Merryville TSR #53, Dog Trap Rd. via Murrumbateman</td>
<td>679682</td>
<td>6128670</td>
<td>YR</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>“Gooroo Hill” Nature Reserve, ACT</td>
<td>691222</td>
<td>6104596</td>
<td>YR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>Justice Robert Hope Memorial Park, ACT</td>
<td>696929</td>
<td>6098444</td>
<td>YR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>“Bobbara Creek”, via Binalong</td>
<td>644035</td>
<td>6165233</td>
<td>YR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>Coolalite TSR #43, Yass – Jerrawa Rd.</td>
<td>680962</td>
<td>6147150</td>
<td>YR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>Bookham Cemetery surrounds, via Bookham</td>
<td>648988</td>
<td>6146239</td>
<td>YR</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>Georges TSR #12, Binda – Wheeo Rd.</td>
<td>710809</td>
<td>6191607</td>
<td>YR</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>McCabes TSR #14, Gunning – Crookwell Rd.</td>
<td>713087</td>
<td>6162880</td>
<td>YR</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>Wargella TSR #28, Yass – Rye Park Rd.</td>
<td>673442</td>
<td>6158130</td>
<td>YR</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>Kangiara TSR #20, Lachlan Valley Way.</td>
<td>660215</td>
<td>6170836</td>
<td>YR</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>Lagoon Creek Roadside Reserve, via Rye Park</td>
<td>676467</td>
<td>6176062</td>
<td>YR</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>McNemey’s TSR #27, Bendenege Travelling Stock Rd.</td>
<td>664384</td>
<td>6158666</td>
<td>YR</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>“Kenmore Dam” site 1 Goulburn – Taralga Rd.</td>
<td>753283</td>
<td>6156637</td>
<td>YR</td>
<td>W</td>
<td>L</td>
</tr>
<tr>
<td>Tidbinbilla Nature Reserve, ACT</td>
<td>673632</td>
<td>6082998</td>
<td>PB</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>“Mullion Creek” Roadside Reserve, Kings Hwy</td>
<td>734214</td>
<td>6095763</td>
<td>PB</td>
<td>S</td>
<td>H</td>
</tr>
<tr>
<td>Black Mountain Nature Reserve, ACT</td>
<td>690114</td>
<td>6093198</td>
<td>PB</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>“Glen Lee”, Dicks Creek Rd. via Murrumbateman</td>
<td>696171</td>
<td>6122686</td>
<td>PB</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>“Gidleigh”, Gidleigh Lane, via Bungendore.</td>
<td>723983</td>
<td>6098936</td>
<td>PB</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>“Lucky Hollow”, Smiths Rd, via Tharwa</td>
<td>692799</td>
<td>6054134</td>
<td>PB</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>Nanimas TSR #50, Nanimas – Murrumbateman Rd.</td>
<td>691372</td>
<td>6123579</td>
<td>PB</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>“Royallea Crown Lease” and “The Scrub”, Royalla</td>
<td>695463</td>
<td>6068494</td>
<td>PB</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>Commissioners Creek TSR, Laggan – Taralga Rd.</td>
<td>745735</td>
<td>6189496</td>
<td>PB</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>Three Mile TSR #2, Crookwell – Binda Rd.</td>
<td>721551</td>
<td>6186489</td>
<td>PB</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>Humes Creek TSR #13, Gunning – Crookwell Rd.</td>
<td>717031</td>
<td>6164314</td>
<td>PB</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>“Mullion Creek”, Kings Hwy</td>
<td>734946</td>
<td>6091015</td>
<td>PB</td>
<td>S</td>
<td>H</td>
</tr>
<tr>
<td>Limekilns Roadside Reserve, Lade Vale Rd.</td>
<td>695205</td>
<td>6137973</td>
<td>PB</td>
<td>L</td>
<td>L</td>
</tr>
<tr>
<td>“Kenmore Dam” site 2 Goulburn – Taralga Rd.</td>
<td>752850</td>
<td>6156826</td>
<td>PB</td>
<td>W</td>
<td>L</td>
</tr>
<tr>
<td>Broadway TSR #30, Blakney Creek Rd</td>
<td>691079</td>
<td>6158060</td>
<td>PB</td>
<td>L</td>
<td>L</td>
</tr>
<tr>
<td>“Farrer Ridge” leasehold Long Gully Lane, ACT</td>
<td>692992</td>
<td>6082008</td>
<td>PB</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>Bullen Range Nature Reserve, ACT</td>
<td>680989</td>
<td>6078022</td>
<td>SR</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>Cavan Woolshed TSR #57, Mountain Creek Rd.</td>
<td>667516</td>
<td>6117715</td>
<td>SR</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>“Glen Lee”, Dicks Creek Rd. via Murrumbateman</td>
<td>695457</td>
<td>6122551</td>
<td>SR</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>“Landsend” leasehold, William Howell Drive. ACT</td>
<td>683889</td>
<td>6097643</td>
<td>SR</td>
<td>M</td>
<td>H</td>
</tr>
<tr>
<td>Brooks TSR #39, Gundaroo – Bungendore Rd.</td>
<td>709720</td>
<td>6113907</td>
<td>SR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>Bruce Ridge Nature Reserve, ACT</td>
<td>690391</td>
<td>6096554</td>
<td>SR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>“Burgess”, via Gearys Gap Lake George</td>
<td>714436</td>
<td>6118395</td>
<td>SR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>Unoccupied crown land, via Geary’s Gap Lake George</td>
<td>715554</td>
<td>6116127</td>
<td>SR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>Alison Hone Reserve, Goulburn – Crookwell Rd.</td>
<td>744417</td>
<td>6160309</td>
<td>SR</td>
<td>W</td>
<td>H</td>
</tr>
<tr>
<td>“Warrambui”, Yass River Rd.</td>
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<td>6130597</td>
<td>SR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>“Belanda Park”, Taylors Flat Rd. via Frogmore</td>
<td>681798</td>
<td>6022747</td>
<td>SR</td>
<td>L</td>
<td>H</td>
</tr>
<tr>
<td>“Wilkes”, Denneys Ln. via Goulburn</td>
<td>750420</td>
<td>6162150</td>
<td>SR</td>
<td>W</td>
<td>H</td>
</tr>
<tr>
<td>Devil’s Elbow Crown Land, Blakney Creek Rd.</td>
<td>690747</td>
<td>6157980</td>
<td>SR</td>
<td>L</td>
<td>L</td>
</tr>
<tr>
<td>Black Mountain Nature Reserve, ACT</td>
<td>691194</td>
<td>6094269</td>
<td>SR</td>
<td>M</td>
<td>L</td>
</tr>
<tr>
<td>“Hillrose”, Kennys Creek Rd. via Boroowra</td>
<td>669301</td>
<td>6188040</td>
<td>SR</td>
<td>L</td>
<td>L</td>
</tr>
<tr>
<td>Nelanglo TSR #48, Gunning – Sutton Rd.</td>
<td>705484</td>
<td>6129410</td>
<td>SR</td>
<td>M</td>
<td>L</td>
</tr>
</tbody>
</table>
Figure 4: At each site 3 plots were located along a stand transect running diagonally across the direction of the main environmental gradient (assumed to be topography), and passing through the centre of the target stand. Plots were evenly distributed along the stand transect (x metres) starting from a random point 1m-100m (y metres) from the stand edge. The distance between plots (x) depended on the length of the stand transect.

Measurements at each plot were based on a 50mx20m (0.1ha) quadrat (Figure 5). The long side of the quadrat was placed parallel to the stand transect so that proportionately more sampling effort occurred in the direction of the main environmental gradient (Figure 4). An 0.1ha quadrat was used because a pilot study indicated this was an appropriate scale at which to estimate basic stand parameters such as mean dbh, basal area, canopy cover, log volume and litter cover. Within the 50mx20m quadrat a 20mx20m sub-quadrat was established for the measurement of vegetation cover and species richness so that these estimates would be consistent with established protocols (e.g. DUAP, 1999; NSW NPWS, 1995). Finally a 50m transect was laid out along the central axis of the 50mx20m quadrat to provide line intersect cover estimates (Walker and Hopkins, 1990) for comparison with the ocular estimates of cover obtained within the 20mx20m sub-quadrat.
Chapter 5: A method for collecting structural attribute data

Figure 5: Measurements at each plot were based on a 50mx20m (0.1ha) quadrat. Each quadrat was laid out from an established starting point X with the long side of the quadrat placed parallel to the stand transect. A 20mx20m sub-quadrat was established within the 50mx20m quadrat for the measurement of vegetation cover and species richness. A 50m transect was also laid out along the central axis of the 50mx20m quadrat, to provide alternative cover measurements using line intersect estimates. 5 litter biomass samples were taken systematically at distances 5, 15, 25, 35 and 45m along the 50m transect at points alternatively 5m either side of the transect.

5.2.2 Measurements taken

Stand level attributes

The measurements taken at each plot were those required to quantify the comprehensive set of stand structural attributes identified in Chapter 3 and summarised in Table 12. Each stand level attribute was quantified as the mean of three plot level estimates.

Plot level measurements

Plot level measurements were conducted using three different sampling methods – 50mx20m quadrat, 20mx20m quadrat and 50m transect. This was done for two reasons: first because convention required a particular method be used for some attributes, for example the use of a 20mx20m quadrat to
estimate species and lifeform richness; and second so that a comparison could be made between the precision and efficiency of estimating the same attribute using different methods, for example visual estimates of lifeform cover using a 20mx20m quadrat could be compared to line intersect estimates using a 50m transect.

These measurements took on average 2 hours to complete, so that three plots or one study site could be assessed in a single day. A summary of the data collected at each plot is provided in Table 15, and described in detail in the following sections. The proforma used to record these field data is included as Appendix 1, and the 75 different structural variables quantified on the basis of this data as Appendix 2. Note that Table 15 is a list of measurements, whereas Table 12 (Chapter 4) is a list of the attributes to be quantified on the basis of these measurements; for example dbh was the measurement (Table 15) used to quantify the attributes, stand basal area and quadratic mean dbh (Table 12).

50MX20M QUADRAT MEASUREMENTS

A 50mx20m (0.1ha) quadrat was used to assess the size and hollow-bearing status of overstorey stems, the size, volume and decay class of coarse woody debris, and the dry weight of litter. This involved recording:

1. For each overstorey stem ≥ 5cm dbh, its species, diameter at breast height (dbh) (within the classes 5-20cm, 20-30cm, 30-40cm, 40-50cm, 50-60cm etc.), whether the stem was dead or alive, and if dead whether it was a stump (<2m tall), and the presence of hollows in three size classes, (small 0-5cm, medium 5-10cm, large >10cm).

2. For each regenerating overstorey stem <5cm dbh, its presence and species.

3. For each piece of downed woody debris (log) >10cm diameter, its length to the nearest 0.5m, average (midpoint) diameter, and decay state within 3 classes (1. Sound when kicked, does not contain cracks, hollows or pipes; 2. Sound when kicked, contains cracks, hollows or pipes; 3. Gives when kicked).

4. For litter, the dry weight of 5 samples taken at points systematically located within the 50mx20m quadrat (Figure 5). At each point all dead organic matter <10cm in diameter was collected within a randomly oriented 0.5mx0.5m quadrat.
Table 15: Summary of measurements taken at each plot. Measurements are grouped under the structural element they address and the sampling method used to collect the data. These data were sufficient to quantify the comprehensive set of attributes identified in Chapter 4, and listed in Table 12.

<table>
<thead>
<tr>
<th>Sampling method</th>
<th>Structural element</th>
<th>Measurements taken</th>
</tr>
</thead>
<tbody>
<tr>
<td>50mx20m quadrat</td>
<td>For each tree stem &gt; 5cm dbh</td>
<td>Species</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Dbh within classes 5-20cm, 20-30cm, 30-40cm, etc.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Dead, alive, or a stump</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Presence of stem or branch hollows in 3 size classes,</td>
</tr>
<tr>
<td></td>
<td>For each regenerating tree stem &lt; 5cm dbh</td>
<td>Presence / absence and species where present</td>
</tr>
<tr>
<td></td>
<td>Stand height</td>
<td>Dominant height based on the mean height of the three tallest trees</td>
</tr>
<tr>
<td></td>
<td>For each piece of downed woody debris &gt;</td>
<td>Length to nearest 0.5m</td>
</tr>
<tr>
<td></td>
<td>10cm diameter</td>
<td>Average diameter measured at midpoint</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Decay state within 3 classes</td>
</tr>
<tr>
<td></td>
<td>Litter &lt; 10cm diameter</td>
<td>Litter biomass based on the mean dry weight of 5 samples</td>
</tr>
<tr>
<td>20mx20m sub-quadrat</td>
<td>Native lifeform diversity</td>
<td>Native lifeform richness using 12 distinct lifeforms</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Native perennial species richness within each lifeform</td>
</tr>
<tr>
<td></td>
<td>Understorey vegetation cover</td>
<td>% Native lifeform cover using projective foliage cover within each lifeform</td>
</tr>
<tr>
<td></td>
<td>Canopy cover</td>
<td>% Combined native and non-native vegetation cover within 4 height classes, 0-0.5m, 0.5-2m, 2-4m, 4-6m</td>
</tr>
<tr>
<td>50m transect</td>
<td>Native lifeform cover</td>
<td>% Native lifeform cover using projective foliage cover within each lifeform based on 10 line intersect samples</td>
</tr>
<tr>
<td></td>
<td>Understorey vegetation cover</td>
<td>% Combined native and non-native vegetation cover within 4 height classes, 0-0.5m, 0.5-2m, 2-4m, 4-6m based on 10 line intersect samples</td>
</tr>
<tr>
<td></td>
<td>Ground cover</td>
<td>% Ground cover of litter, cryptogams, bare earth, rocks and logs based on 10 x 1m line intersect samples</td>
</tr>
<tr>
<td></td>
<td>Canopy cover</td>
<td>% Canopy projective cover based on 50 canopy samples using an optical crown-o-meter</td>
</tr>
</tbody>
</table>
Chapter 5: A method for collecting structural attribute data

20MX20M SUB-QUADRAT MEASUREMENTS

A 20mx20m (0.04ha) quadrat was used to assess the richness of native lifeforms and perennial species, and the projective foliage cover of native and non-native lifeforms. This involved recording:

1. The number of native lifeforms, using 12 distinct lifeforms (tussock grass, non-tussock grass, low shrub 0-0.5m, tall shrub > 0.5m, sedges / rushes, ferns, vines, xanthorrhoea, mistletoe, low regeneration < 2m, tall regeneration > 2m, tree), based on lifeforms defined by Walker and Hopkins (1990).
2. The number of perennial native species within each lifeform; annual species were not assessed because plots were measured at different times of the year, and during an extended period of drought conditions.
3. The projective foliage cover within each native lifeform, using ocular estimates to the nearest 1%.
4. The combined projective foliage cover of native and non-native vegetation within 3 height classes, 0-0.5m, 0.5-2m, 2-6m, using ocular estimates to the nearest 1%.

50M TRANSECT MEASUREMENTS

The 50m transect was used to estimate ground cover, and to obtain alternative line intersect estimates of the cover of native and non-native vegetation for comparison with ocular estimates made using the 20mx20m quadrat. This involved:

1. 10 line intersect estimates of cover using 1m intervals equally spaced along the 50m transect (0-1m, 5-6m, 10-11m, 15-16m, etc.). Over each interval % cover was estimated to the nearest 5%. These ten estimates were then used to estimate mean values over the transect to the nearest 1% for:
   - Projective foliage cover within each native lifeform;
   - Combined projective foliage cover of native and non-native vegetation within 3 height classes, 0-0.5m, 0.5-2m, 2-6m;
   - % ground cover of litter, cryptogams, bare earth, rocks and logs.

10 line intersect estimates were used because a pilot study indicated that this sampling effort produced stable means for both litter and ground vegetation cover.

2. Estimating canopy projective cover by systematically sampling every metre...
along the 50m transect with a crown-o-meter periscope (Heatherington, 1967),
an optical device with cross hairs that sights vertically into the canopy. At each
sample point a score of 1 was awarded if the cross hairs were coincident with a
canopy leaf or branch, otherwise the score was 0. Canopy projective cover was
the mean of the 50 scores.

In the next chapter I present the results of a three-stage analysis of the data
that were collected using the methods described above. Through this analysis
the comprehensive suite of structural attributes identified in Chapter 5, is
systematically reduced to a core set of 13 attributes.
Chapter 6: Arriving at a core set of attributes for characterising structural complexity in dry sclerophyll forests and woodlands of south-eastern Australia

Chapter summary

This chapter presents the results of a three-stage analysis of data collected using the methods detailed in Chapter 5. The first stage – a preliminary analysis – indicated that the 48 study sites represented a broad range of condition, and that the two dry sclerophyll communities could be treated as a single community, which was structurally distinct from the woodland community. In the second stage of the analysis, thirteen core attributes were identified using the criteria that a core attribute should:
1. Be either evenly or approximately normally distributed amongst study sites;
2. Distinguish between woodland and dry sclerophyll communities;
3. Function as a surrogate for other attributes;
4. Be efficient to measure in the field.

The core attributes were: Vegetation cover <0.5m; Vegetation cover 0.5-6.0m; Perennial species richness; Lifeform richness; Stand basal area of live trees; Quadratic mean diameter of live stems; ln(number of regenerating stems ha\(^{-1}\)+1); ln(number of hollow-bearing trees ha\(^{-1}\)+1); ln(number of dead trees ha\(^{-1}\)+1); (number of live stems ha\(^{-1}\) > 40cm dbh); (total log length ha\(^{-1}\)); (total large log length ha\(^{-1}\)); Litter dry weight.

The analysis also demonstrated that the 13 core attributes could be modelled as continuous variables, and that these variables were indicative of the scale at which the different attributes operated.

In the third and final stage of the analysis, Principal Components Analysis was used to test for redundancy amongst the core attributes by representing core attributes as vectors showing the relative importance of each attribute in explaining variation between sites, and the degree of correlation between attributes. Although this analysis highlighted six groupings within which attributes were correlated to some degree, these relationships were not considered sufficiently robust to justify reducing the number of core attributes.
6.1 Introduction

This chapter addresses stage four of the methodology for developing an index of structural complexity outlined in Chapter 3. The aim of this stage was to establish a subset of attributes – a core set – from the comprehensive set of structural attributes identified in Chapter 4. The rationale for identifying a core set was that these attributes could then form an efficient basis for constructing an index of structural complexity. Core attributes were therefore considered to be those that best distinguished between study sites, and which could function as surrogates for other attributes.

A three-stage analysis was used to identify core attributes from the comprehensive suite. This analysis utilised data collected from 144 plots at 48 study sites in a woodland and two dry sclerophyll forest communities, as described in Chapter 5. In the first stage, a preliminary analysis was used to check that these 48 study sites represented a broad range of vegetation condition states, and to assess whether the three vegetation types should be treated as structurally distinct communities. In the second stage, univariate relationships were used to identify potential core attributes on the basis of four criteria. In the third and final stage, Principal Components Analysis was used to test for redundancy amongst the core attributes.

6.2 Preliminary analysis

As a preliminary analysis, mean values and ranges for 12 structural attributes were quantified in each of the three vegetation communities. I also tested for significant differences in these means between the three communities (ANOVA, students t test for comparing pairs of means) (Table 16). These particular 12 attributes were chosen for the preliminary analysis because their importance was well established in the literature (e.g. Table 1, Chapter 2). The number of cut stumps was included on the grounds that it was indicative of the degree of modification since European settlement. At each site attributes were quantified as the stand level mean of three plot level estimates, because by definition a stand has an average structural condition that differentiates it from adjacent areas (Lindenmayer and Franklin, 2002). There were 16 study sites in each
vegetation community, so that the mean values and ranges for the 12 attributes were indicative of the 16 different states of vegetation condition sampled for each community.

Table 16: Mean values and ranges (n=16), for a selection of structural attributes in one woodland and two dry sclerophyll forest communities. Attributes were quantified using data from 144 plots in 48 study sites. Superscripts indicate significant (p < 0.05) differences between means (e.g. ¹ indicates a mean value which is significantly different to the mean for community 1, the Yellow Box–Red Gum community). NS indicates no significant difference to means in either of the other two communities.

<table>
<thead>
<tr>
<th>Structural attributes</th>
<th>Woodland</th>
<th>Dry sclerophyll forest</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cut stumps (ha⁻¹)</td>
<td>Mean</td>
<td>Range</td>
</tr>
<tr>
<td>Stand height (m)</td>
<td>20 ²,³  17 - 24</td>
<td>17 ¹  12 - 20</td>
</tr>
<tr>
<td>Projective canopy cover (%)</td>
<td>27 ²,³  8 - 57</td>
<td>42 ¹  36 - 60</td>
</tr>
<tr>
<td>Basal area live trees (m²ha⁻¹)</td>
<td>17 ²,³  8 - 47</td>
<td>28 ¹  15 - 42</td>
</tr>
<tr>
<td>Quadratic mean dbh (cm)</td>
<td>38 ²,³  20 - 78</td>
<td>28 ¹  18 - 37</td>
</tr>
<tr>
<td>Hollow-bearing trees (ha⁻¹)</td>
<td>9 ²,³  0 - 43</td>
<td>51 ¹  3 - 150</td>
</tr>
<tr>
<td>Dead trees (ha⁻¹)</td>
<td>17 ²,³  0 - 67</td>
<td>76 ¹  10 - 213</td>
</tr>
<tr>
<td>Perennial species (per 400m²)</td>
<td>13 ²,³  5 - 27</td>
<td>19 ¹  13 - 27</td>
</tr>
<tr>
<td>Vegetation cover &lt;0.5m (%)</td>
<td>54 ²,³  18 - 91</td>
<td>35 ¹  11 - 56</td>
</tr>
<tr>
<td>Vegetation cover &gt;0.5m (%)</td>
<td>2 ²,³  0.1 – 6</td>
<td>4 ¹  1 – 8</td>
</tr>
<tr>
<td>Litter cover (%)</td>
<td>59 ²,³  30 - 78</td>
<td>77 ¹  66 - 91</td>
</tr>
<tr>
<td>Log volume (m³ha⁻¹)</td>
<td>11 ²  0.2 - 24</td>
<td>20 ¹  7 - 52</td>
</tr>
</tbody>
</table>

The results of the preliminary analysis indicated that within each community a considerable range of variation had been sampled for each structural attribute. The range and mean number of cut stumps also confirmed that each community contained sites representing a variety of levels of modification by Europeans. On this basis, I concluded that the study sites represented a broad suite of condition states for each of the three different communities.

There were no significant differences between the two dry sclerophyll communities for any of the 12 attributes, whereas the woodland community was
significantly different to the Broadleaved Peppermint–Brittle Gum community for 11 of the 12 attributes, and to the Scribbly Gum–Red Stringybark community for 10 of the 12 attributes. On this basis, the two dry sclerophyll communities were treated as a single community in subsequent stages of the analysis.

6.3 Core attributes identified from univariate analysis

6.3.1 Criteria for identifying core attributes

In the second stage of the analysis, univariate relationships were used to identify potential core attributes on the basis of the following four criteria:

1. The attribute should be either evenly or approximately normally distributed amongst study sites. A highly skewed distribution would indicate that many sites have similar values for the attribute, which would limit its potential to distinguish between sites. An even distribution would maximise discrimination, although this distribution was rarely observed in attributes. In some cases, attributes with a skewed distribution could be normalised using an appropriate transformation, or by combining attributes which described a common structural element (e.g. coarse woody debris) in a simple index.

2. The attribute should distinguish between woodland and dry sclerophyll communities. This criterion reflected the results of the preliminary analysis, which demonstrated that for a range of important attributes there were significant differences between attribute levels in woodland and dry sclerophyll communities. Therefore if an attribute was not useful for discriminating between these two structurally distinct communities it was unlikely to be useful for discriminating sites within each of these communities - this is because the differences between communities will be more pronounced than the differences within communities.

3. The attribute should function as a surrogate for other attributes. Correlations with other attributes in which Pearson’s correlation coefficient r exceeded 0.60 (p<0.001), were indicative of this criterion.

4. The attribute should be simple to measure in the field. Where two attributes were correlated the one that was easiest to measure, and most instructive for management, was preferred.

Thirteen core attributes were identified from an initial suite of 75 (Appendix 2)
using these four criteria. These core attributes and other correlated attributes ($r > 0.60$) are shown in Table 17. The results for the 13 core attributes and correlates are then described in detail.

### 6.3.2 Understorey vegetation cover: vegetation cover $<0.5$m and vegetation cover $0.5$-$6.0$m

Understorey vegetation cover was measured for each native lifeform, and within three height classes for native and non-native vegetation combined. Estimates were made using two different methods – ocular estimates within a 20mx20m quadrat, and line intersect estimates systematically placed along a 50m transect. With the exceptions of the height class 2.0-6.0m, and the lifeform tall regeneration, both of which had very low levels of cover, the two methods were in close agreement (Table 18). This confirmed the validity of making ocular estimates to the nearest 1% rather than using cover classes, such as those developed by Braun Blanquet (Mueller-Dombois and Ellenberg, 1974) – a conclusion supported by Austin et al. (2000). Ocular estimates were also more efficient to use in the field than the line intersect method, and were therefore adopted as the preferred method for estimating cover. All subsequent analysis in this chapter is based on data using ocular estimates of cover.

With the exception of tussock grasses, understorey lifeforms were present at low levels of cover (Table 18), so that vegetation cover was better characterised using the two height classes, $<0.5$m and $0.5$-$6.0$m, rather than cover within separate lifeforms. Non-indigenous vegetation cover was included in these height classes because it can contribute resources that are important for faunal diversity (e.g. Evans, 2000; French and Zubovich, 1997). These two height classes met the four criteria for selection as core attributes.
Table 17. Thirteen core structural attributes identified from an initial suite of 75 (Appendix 2) using the criteria that a core attribute should: 1. Be either evenly or approximately normally distributed amongst study sites; 2. Distinguish between woodland and dry sclerophyll communities; 3. Function as a surrogate for other attributes; 4. Be efficient to measure in the field. Only correlations in which Pearson’s r > 0.60 were considered as correlates. (* significantly different means (p<0.05). Dry sclerophyll forest. ^ Original measurement units)

<table>
<thead>
<tr>
<th>Core attribute</th>
<th>Means</th>
<th>Correlates (r &gt; 0.60)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vegetation cover &lt;0.5m (%)</td>
<td>Woodland</td>
<td>^DSF</td>
</tr>
<tr>
<td>Vegetation cover 0.5-6.0m (%)</td>
<td>2.3*</td>
<td>4.4*</td>
</tr>
<tr>
<td>Perennial species richness (per 400m²)</td>
<td>12.5*</td>
<td>20.1*</td>
</tr>
<tr>
<td>Lifeform richness (per 400m²)</td>
<td>6.3*</td>
<td>8.3*</td>
</tr>
<tr>
<td>Stand basal area of live trees (m²ha⁻¹)</td>
<td>16.8*</td>
<td>27.0*</td>
</tr>
<tr>
<td>Quadratic mean diameter of live stems (dbh_Q) (cm)</td>
<td>37.9*</td>
<td>24.8*</td>
</tr>
<tr>
<td>ln(number of regenerating stems ha⁻¹+1)</td>
<td>149^B</td>
<td>177^B</td>
</tr>
<tr>
<td>ln(number of hollow-bearing trees ha⁻¹+1)</td>
<td>5.2*B</td>
<td>27.4*B</td>
</tr>
<tr>
<td>ln(number of dead trees ha⁻¹+1)</td>
<td>7.3*B</td>
<td>50.5*B</td>
</tr>
<tr>
<td>(number of live stems ha⁻¹ &gt; 40cm dbh)</td>
<td>32^B</td>
<td>35.7^B</td>
</tr>
<tr>
<td>(total log length ha⁻¹) (mha⁻¹)</td>
<td>182*B</td>
<td>519*B</td>
</tr>
<tr>
<td>(total large log length ha⁻¹) (mha⁻¹)</td>
<td>34.5^B</td>
<td>35.7^B</td>
</tr>
<tr>
<td>Litter dry weight (tha⁻¹)</td>
<td>6.8*</td>
<td>14.3*</td>
</tr>
</tbody>
</table>
Table 18. Mean projective foliage cover of perennial vegetation within different lifeforms and height classes. Cover was quantified using ocular estimates of cover in 20mx20m quadrats, and 10 line intersect estimates along a 50m transect. With the exceptions of the height class 2.0-6.0m, and the lifeform tall regeneration, both methods were highly correlated. This confirmed the validity of making ocular estimates to the nearest 1%, rather than within cover classes. Note: the lifeform forbs was not assessed because the majority of forb species could not be reliably detected in all seasons or years.

<table>
<thead>
<tr>
<th>Lifeform / class</th>
<th>Ocular estimate (%)</th>
<th>Line intersect estimate (%)</th>
<th>Correlation (Pearson’s r)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0 – 0.5m</td>
<td>37.3</td>
<td>36.3</td>
<td>0.92</td>
</tr>
<tr>
<td>0.5 –2.0</td>
<td>3.1</td>
<td>3.3</td>
<td>0.75</td>
</tr>
<tr>
<td>2.0 – 6.0</td>
<td>0.58</td>
<td>2.19</td>
<td>0.19</td>
</tr>
<tr>
<td>Tussock grass</td>
<td>27</td>
<td>22.9</td>
<td>0.86</td>
</tr>
<tr>
<td>Shrub low</td>
<td>1.9</td>
<td>3.0</td>
<td>0.88</td>
</tr>
<tr>
<td>Shrub tall</td>
<td>2.2</td>
<td>3.4</td>
<td>0.75</td>
</tr>
<tr>
<td>Sedge rush</td>
<td>1.68</td>
<td>2.6</td>
<td>0.87</td>
</tr>
<tr>
<td>Fern</td>
<td>0.08</td>
<td>0.05</td>
<td>0.84</td>
</tr>
<tr>
<td>Vine</td>
<td>0.05</td>
<td>0.07</td>
<td>0.55</td>
</tr>
<tr>
<td>Low regeneration</td>
<td>0.82</td>
<td>0.88</td>
<td>0.70</td>
</tr>
<tr>
<td>Tall regeneration</td>
<td>0.58</td>
<td>0.67</td>
<td>0.29</td>
</tr>
<tr>
<td>Tree</td>
<td>37</td>
<td>40</td>
<td>0.81</td>
</tr>
</tbody>
</table>

In addressing the four criteria for core attributes, vegetation cover <0.5m:

1. Was normally distributed amongst study sites, and ranged from a minimum of 3.2% to a maximum of 91.0%, with a mean value of 37.3%;
2. Distinguished between woodland and dry sclerophyll communities, with woodland sites having significantly more vegetation cover <0.5m (54.2%) than dry sclerophyll sites (28.8%);
3. Was correlated with the cover of tussock grasses (r = 0.72), the lifeform which dominated this height class;
4. Was straightforward to measure in the field because this height class represented a clearly defined stratum.

In addressing the four criteria for core attributes, vegetation cover 0.5-6.0m:

1. Was approximately normally distributed amongst study sites, ranging
from a minimum of 0.1% to a maximum of 9.5%, with a mean of 3.7%;
2. Distinguished between woodland and dry sclerophyll communities, with significantly more vegetation cover >0.5m in dry sclerophyll sites (4.4%) than in woodland sites (2.3%);
3. Was highly correlated with tall shrub cover (r = 0.91), the lifeform which dominated this height class, and with shrub species richness (r = 0.64), although this correlation was not as strong because low shrubs did not contribute to this height class;
4. Was straightforward to measure because most of the cover was attributable to tall shrubs, which occupied a clearly defined stratum.

6.3.3 Compositional diversity: Lifeform richness and species richness

Diversity is commonly defined in terms of two factors, richness and relative abundance; for example lifeform diversity can be quantified in terms of lifeform richness and foliage cover within each lifeform (Magurran, 1988). For the communities measured in this study, most lifeforms were present at very low levels of cover, although the number of indigenous species within individual lifeforms could be high. Compositional diversity was therefore better described by the richness of indigenous perennial species within lifeforms rather than by the abundance of cover within lifeforms.

This type of compositional diversity (C') was quantified by applying the Shannon-Weiner Index (Magurran, 1988) in the form $C' = -\sum p_i \ln p_i$, where $p_i$ is the proportion of perennial species belonging to the ith lifeform. Across the study sites, $C'$ ranged from a minimum of 0.98 to a maximum of 2.04, with significantly higher lifeform diversity in dry sclerophyll sites (1.83) compared with woodland sites (1.49). $C'$ was highly correlated with lifeform richness (r = 0.97), and lifeform richness was in turn correlated with perennial species richness (r = 0.75). This suggested that lifeform richness could be an efficient surrogate for compositional diversity. However, when only dry sclerophyll sites were considered the correlation between lifeform richness and perennial species richness was weak (r = 0.38). This was because in dry sclerophyll sites two lifeforms, low shrubs and tall shrubs, contributed a disproportionately large amount to perennial species richness. It was concluded that compositional
diversity was best described using perennial species richness in addition to lifeform richness, both of which met the four criteria for selection as core attributes.

In addressing the four criteria, indigenous perennial species richness:

1. Was approximately normally distributed amongst study sites, ranging from a minimum of 4.7 to a maximum of 27.3, with a mean value of 17.6 species per 400m$^2$ quadrat;
2. Distinguished between woodland and dry sclerophyll communities with significantly more perennial species in dry sclerophyll sites (20.1) than in woodland sites (12.5);
3. Was correlated with shrub species richness ($r = 0.85$), lifeform richness ($r = 0.75$), and compositional diversity $C'$ ($r = 0.72$); it was also negatively correlated with the quadratic mean diameter of live trees $dbh_Q$ ($r = -0.61$), and the standard deviation of $dbh_Q$ ($r = -0.63$), reflecting an association between a large standard deviation of $dbh_Q$ and a large mean $dbh_Q$ in woodland sites;
4. Was straightforward to measure because it excluded species that are not visible in all seasons or years, and did not require exact species identification, or estimates of species abundance.

In addressing the four criteria, indigenous lifeform richness:

1. Was approximately normally distributed and ranged from a minimum of 3.0 to a maximum of 9.6, with a mean of 7.6 lifeforms per 400m$^2$ quadrat;
2. Distinguished between woodland and dry sclerophyll communities with significantly more lifeforms present in dry sclerophyll sites (8.3) than in woodland sites (6.3);
3. Was correlated with compositional diversity $C'$ ($r = 0.97$), perennial species richness ($r = 0.75$), and shrub species richness ($r = 0.61$), and like perennial species richness was negatively correlated with the mean $dbh_Q$ of live trees ($r = -0.64$), and the standard deviation of $dbh_Q$ ($r = -0.62$);
4. Was straightforward to measure because the identification of different lifeforms did not require specialist botanical knowledge.
6.3.4 Canopy cover and stand basal area

Canopy projective cover (canopy cover) was estimated using two different methods – ocular estimates within a 20mx20m quadrat, and crown-o-meter estimates at 50 points systematically placed along a 50m transect. The two methods were in close agreement (r = 0.81). This confirmed the validity of making ocular estimates to the nearest 1% and treating canopy cover as a continuous variable rather than using cover classes, which would lose information and mask differences between sites. Using data from ocular estimates, canopy cover was normally distributed amongst sites, ranging from a minimum of 8% to a maximum of 62% with a mean of 37%. Dry sclerophyll sites had significantly more canopy cover (42%) than woodland sites (27%) although the range of canopy cover was similar for both communities – dry sclerophyll sites 13-62% and woodland sites 8-57%.

Canopy cover was correlated with stand basal area (r = 0.77), reflecting a well-established relationship between tree dbh and crown diameter (Lane Poole, 1936). Stand basal area met the four criteria for a core attribute and was selected in preference to canopy cover, because it was easier to measure precisely, and was correlated with a greater number of other attributes.

In addressing the four criteria, stand basal area:

1. Was normally distributed amongst study sites, ranging from a minimum of 8.0 m²ha⁻¹ to a maximum of 47.1 m²ha⁻¹, with a mean value of 23.6 m²ha⁻¹;
2. Distinguished between woodland and dry sclerophyll communities with significantly higher stand basal area occurring in dry sclerophyll sites (27.0 m²ha⁻¹) than in woodland sites (16.8 m²ha⁻¹), although the range was similar for both communities — dry sclerophyll sites 8.6-42.2 m²ha⁻¹ and woodland sites 8.0-47.1 m²ha⁻¹;
3. Was correlated with canopy cover (r = 0.77), ln(number of hollow-bearing trees+1) (r = 0.68), ln(number of dead trees+1) (r = 0.61), √(length of logs) (r = 0.67) and litter dry weight (r = 0.61); these correlations reflected associations between increasing stand basal area and the presence of large mature trees containing hollows (Gibbons et al., 2000, Braithwaite
et al., 1983), the expression of density dependent mortality (Oliver and Larson, 1996), the production of coarse woody debris from dead trees and the disintegrating crowns of mature trees, and increasing canopy foliage which was the main source of litter in these grassy forest ecosystems.

4. Was straightforward to measure.

6.3.5 The number and size of live trees: quadratic mean diameter

The number of live trees was estimated from a count of all live stems ≥ 5cm dbh within a 0.1 ha plot. Stems with a dbh < 5cm were considered to be regeneration and were not included. The number of live trees ranged from a minimum of 17 stems ha⁻¹ to a maximum of 2067 stems ha⁻¹, with a mean of 507 stems ha⁻¹. The ln(number of live trees) was normally distributed with significantly fewer live stems in woodland sites (155 stems ha⁻¹) compared with dry sclerophyll sites (571 stems ha⁻¹).

The average size and variability in size of live trees were estimated by measuring the dbh of all live stems > 5cm dbh in a 0.1 ha plot. Average size was quantified in terms of the quadratic mean dbhₐ – where dbhₐ = \sqrt{\frac{(\sum \text{dbh}^2)}{N}} – rather than the arithmetic mean. This was because dbhₐ (cm) is directly related to stand basal area (BA m²ha⁻¹) and the number of live trees (N stemsha⁻¹) through the relationship: BA = \frac{\pi}{4} \times (dbhₐ/200)^2 \times N

Consequently dbhₐ is a good surrogate for the average number of live trees, in addition to representing the average size of trees. This was confirmed by the strong correlation between dbhₐ and ln(number of live trees) (r = -0.88).

The standard deviation of dbhₐ was used to quantify variation in tree size; an approach supported by a number of different studies (e.g. Neumann and Starlinger, 2001; Acker et al., 1998; Van Den Meerschautt and Vandekerkhove, 1998; Spies and Franklin, 1991). The standard deviation of dbhₐ was normally distributed amongst study sites and ranged from a minimum of 2.2cm to a maximum of 39.8cm with a mean of 15.5cm. It differentiated between woodland and dry sclerophyll sites, with woodland sites (22.0cm) having significantly more variation in tree size than dry sclerophyll sites (12.2cm). The standard deviation
of \(dbh_Q\) was highly correlated with \(dbh_Q\) \((r = 0.87)\). \(dbh_Q\) was therefore adopted as a surrogate for both the standard deviation of \(dbh_Q\) and the \(\ln(\text{number of live trees})\). It also met all four criteria for selection as a core attribute because it:

1. Was approximately normally distributed across study sites and ranged from a minimum of 13.1cm to a maximum of 78.0cm, with a mean of 29.2cm;
2. Differentiated between woodland and dry sclerophyll communities, with woodland sites having significantly larger trees (37.9cm) than dry sclerophyll sites (24.8cm);
3. Was correlated with \(\ln(\text{number of live trees})\) \((r = -0.88)\), and the standard deviation of \(dbh_Q\) \((r = 0.87)\);
4. Was efficient to measure because stems were assessed on the basis of \(dbh\) classes (5-20, 20-30, 30-40 etc) rather than exact \(dbh\) measurements, and on average each 0.1 ha plot contained no more than 50 live stems.

### 6.3.6 Regeneration

I limited the direct assessment of regeneration to long-lived overstorey species. This was because the direct assessment of understorey species regeneration can be difficult in practice, due to its small size, sparse cover and often similar growth form to adult plants. I decided that the most efficient way to monitor understorey regeneration was indirectly through its effects on vegetation cover, lifeform richness and perennial species richness.

Overstorey regeneration was defined as all live stems with a \(dbh < 5\)cm. These were quantified in terms of visual estimates of the cover of low (0-2m) and tall (>2m) regenerating stems, and also the number of regenerating stems \(\text{ha}^{-1}\) based on a count of regenerating stems in an 0.1ha plot. The timing of these assessments in relation to past disturbance was not considered critical, because in woodland and dry sclerophyll communities overstorey regeneration can accumulate continuously as lignotubers (Florence, 1996).

Tall and low regeneration were present at very low levels of cover. The cover of tall regeneration ranged from 0-1.8% with a mean of 0.6%, and the cover of low
regeneration ranged from 0-3.9% with a mean of 0.8%. Both categories of cover were correlated with the number of regenerating stems ha\(^{-1}\), which was considered to be a more robust measure of regeneration because it was a count that could be precisely measured in the field. The number of regenerating stems ranged from a minimum of 14 stems ha\(^{-1}\) to a maximum of 1557 stems ha\(^{-1}\). It had a skewed distribution with more than 50% of sites having less than 170 stems ha\(^{-1}\). However, the ln(number of regenerating stems ha\(^{-1}\)+1) was normally distributed and was selected as a core attribute on the basis that it met 3 of the 4 criteria for selection, and quantified a key functional process (McIntyre et al., 2002).

In addressing the four criteria, ln(number of regenerating stems ha\(^{-1}\)+1):

1. Was normally distributed;
2. Did not differentiate between woodland and dry sclerophyll communities, with similar levels of overstorey regeneration in woodland (149 stems ha\(^{-1}\)) and dry sclerophyll sites (177 stems ha\(^{-1}\));
3. Was correlated with both the cover of low regeneration (r = 0.78) and the cover of tall regeneration (r = 0.60);
4. Was efficient to measure because the size and distinct growth form of regeneration made it easy to identify in the field, and 75% of sites had less than 35 regenerating stems present in a 0.1ha plot.

### 6.3.7 Hollow-bearing trees

The number of hollow-bearing trees was quantified by counting the number of hollow-bearing trees, dead or alive, in a 0.1ha plot. The number of hollow-bearing trees ranged from 0 to 150 (stems ha\(^{-1}\)) and had a skewed distribution with more than 50% of sites having less than 20 hollow-bearing trees ha\(^{-1}\). The diameter distribution of hollow-bearing trees across all study sites, indicated that most hollow-bearing trees occurred in the smaller diameter classes (Figure 6). However because there were more trees present in the smaller diameter classes, the probability that any individual stem would contain a hollow decreased with diameter (Figure 7). The probability of the presence of a hollow was modelled using logistic regression (Legendre and Legendre, 1998).
Figure 6. Most hollow-bearing trees occurred in the smaller diameter classes, as indicated by the diameter distribution of hollow-bearing trees across all study sites. The distribution shows the total count of hollow trees recorded in the 16 woodland and 32 dry sclerophyll study sites.
Figure 7. Larger trees were more likely to contain hollows than smaller trees as indicated by the relationship between the probability that a tree contained a hollow and tree diameter, using data from 10,634 trees assessed in 16 woodland and 32 dry sclerophyll sites. Logistic regression was used to model the occurrence of hollows because the response variable was in the form of presence / absence data (Legendre and Legendre, 1998). Probability of a tree containing a hollow = 1/(1+exp(4.5150-0.060(dbh)))  \( p < 0.0001 \) where dbh is measured in cm.
Hollow-bearing trees were also assessed in terms of the presence of hollows in three size classes, small 0-5cm, medium 5-10cm and large >10cm. This was done because different arboreal species utilise different sized hollows, both in terms of preferred entry size and internal hollow dimensions (Whitford, 2001; Lindenmayer and Franklin, 1997; Gibbons and Lindenmayer, 1996), so that maximum diversity of arboreal fauna requires a range of hollow sizes.

Hollow diversity was considered to be a measure of the relative abundance of hollows occurring in the three size classes. This was quantified by constructing a Hollow Diversity Index, which scored sites according to the number of trees containing hollows in each size class. In this approach the same tree could contribute hollows to more than one size class. To construct the index, abundance classes were defined on the basis of the quartiles of the distribution of hollow-bearing trees for each hollow size. These abundance classes were then awarded a score from 0-3 for each hollow size (Table 19). The Hollow Diversity Index was the sum of the scores for each hollow size class. This gave a maximum score of 9, for sites in which the number of hollow-bearing trees exceeded the 75th percentile in all three size classes, and a minimum score of 0, for sites in which the number of hollow-bearing trees was less than the 25th percentile in all 3 size classes.

Table 19. Scoring a Hollow Diversity Index on the basis of hollow-bearing tree abundance (stems ha$^{-1}$) in three hollow size classes (small < 5cm, medium 5-10cm and large > 10cm). For each size class a score from 0-3 was awarded by comparing hollow-bearing tree abundance to the quartiles of the distribution of hollow-bearing trees for that hollow size class. The maximum score on the index was 9 and the minimum 0.

<table>
<thead>
<tr>
<th>Quartile</th>
<th>Stems ha$^{-1}$ with small hollows (&lt; 5cm)</th>
<th>Stems ha$^{-1}$ with medium hollow (5-10cm)</th>
<th>Stems ha$^{-1}$ with large hollows (&gt; 10cm)</th>
<th>Score</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-25%</td>
<td>0-3.3</td>
<td>0 – 1.8</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>25-50%</td>
<td>3.3 – 13.4</td>
<td>1.8 – 6.4</td>
<td>0 – 3.3</td>
<td>1</td>
</tr>
<tr>
<td>50-75%</td>
<td>13.4 – 32.5</td>
<td>6.4 – 17.7</td>
<td>3.3 – 13.3</td>
<td>2</td>
</tr>
<tr>
<td>75-100%</td>
<td>&gt;32.5</td>
<td>&gt; 17.7</td>
<td>&gt;13.3</td>
<td>3</td>
</tr>
</tbody>
</table>
The Hollow Diversity Index was approximately normally distributed and ranged from 0 to 9 with a mean of 4.7. There was significantly higher Hollow Diversity in dry sclerophyll sites (5.5) compared to woodland sites (3.1). The Hollow Diversity Index was highly correlated with ln(number of hollow-bearing trees ha\(^{-1}\)+1) (r = 0.89), which was an attribute that better met the criteria for selection as a core attribute. This was because it only required hollows to be assessed on an absence / presence basis rather than in terms of three different size classes.

In addressing the four criteria, ln(number of hollow-bearing trees ha\(^{-1}\)+1):
1. Was normally distributed;
2. Differentiated between woodland and dry sclerophyll communities with significantly more hollow-bearing trees occurring in dry sclerophyll sites (27.4 stems ha\(^{-1}\)) compared to woodland sites (5.2 stems ha\(^{-1}\));
3. Was correlated with the Hollow Diversity Index (r = 0.89), stand basal area (r = 0.68), ln(number of dead trees ha\(^{-1}\)+1) (r = 0.62) and (total log length) (r = 0.62);
4. Was straightforward to measure in the field, because 75% of sites contained less than 5 hollow-bearing trees in a 0.1 ha plot

6.3.8 Number of dead trees

Dead trees contained a disproportionate share of the hollow-bearing trees occurring in the study sites. Although only 9.8 % of all stems were dead trees they accounted for 23.2% of all hollow-bearing stems, 26.8% of all trees with small hollows, 17.8% of all trees with medium hollows and 20.2% of all trees with large hollows. These results support those of Gibbons et al. (2002), who found that dead trees accounted for 18.5% of all trees with hollows in temperate forests in south-eastern Australia.

In terms of basal area, dead trees on average equated to 7% of the basal area of live trees. There was a significantly larger basal area of dead trees in dry sclerophyll sites (1.4 m\(^2\)ha\(^{-1}\)) compared to woodland sites (0.4 m\(^2\)ha\(^{-1}\)), and in terms of average dbh dead trees in dry sclerophyll sites were also significantly larger (17.2 cm) than those in woodland sites (12.0 cm). These results may reflect the past removal of large dead trees from woodland sites, because these
sites are more intensively managed than dry sclerophyll sites and contain tree species that are valued as firewood (e.g. Driscoll et al., 2000).

Across the study sites the number of dead trees ranged from a minimum of 0 to a maximum of 460.0 stems ha\(^{-1}\) with a mean of 60.8 stems ha\(^{-1}\). The distribution of the number of dead trees was skewed with more than 50% of sites having less than 34 dead stems ha\(^{-1}\). The \(\ln(\text{number of dead trees}+1)\) was normally distributed and met all four criteria for selection as a core attribute.

In addressing the four criteria, \(\ln(\text{number of dead trees}\ ha^{-1}+1)\):

1. Was normally distributed;
2. Differentiated between woodland and dry sclerophyll communities with significantly more dead trees in dry sclerophyll sites (50.5 stems ha\(^{-1}\)) than in woodland sites (7.3 stems ha\(^{-1}\));
3. Was correlated with the basal area of dead trees \((r = 0.65)\), \(\ln(\text{number of live stems}\ ha^{-1})\) \((r = 0.68)\), canopy cover \((r = 0.66)\) and stand basal area \((r = 0.61)\) reflecting the expression of density-dependent mortality with increasing stand density, and with the \(\ln(\text{number of hollow-bearing trees}\ ha^{-1}+1)\) \((r = 0.62)\) reflecting the disproportionate number of hollow trees which were dead trees;
4. Was straightforward to measure because 75% of sites contained less than 7 dead trees in a 0.1 ha plot.

**6.3.9 Number of large trees**

The number of large trees was defined as the number of live stems ha\(^{-1}\) with a dbh > 40cm. A threshold of 40cm was used because in dry sclerophyll and woodland communities stems of this size are likely to be established mature trees, either containing hollows or enduring to develop them in the future; for example 44% of all hollow-bearing trees had a dbh >40cm, this included 78% of all stems with large hollows, 64% of all stems with medium hollows and 38% of all stems with small hollows. Raising the threshold to a larger dbh was considered inappropriate because this reduced the ability to distinguish between sites with this attribute, with many sites recording zero for the number of large trees; for example only 4% of sites had no trees with dbh > 40cm,
whereas 20% of sites had no trees with dbh > 60cm.

The number of live stems ha\(^{-1}\) > 40cm dbh ranged from a minimum of 0 to a maximum of 123, and had a skewed distribution with more than 25% of study sites having less than 20 live stems ha\(^{-1}\) with a dbh > 40cm. However the (number of live stems ha\(^{-1}\) > 40cm dbh) was normally distributed amongst study sites and was selected as a core attribute despite meeting only two of the four criteria for selection. This decision reflected the established functional importance of large trees (e.g. Franklin et al., 1981), and the failure to identify a suitable surrogate, which operated in both dry sclerophyll and woodland communities. For example, the number of large trees was relatively weakly correlated with stand basal area (r = 0.56) and quadratic mean dbh (r = 0.38). This reflected the past removal of much of the overstorey in woodland sites, so that these sites were now characterised by a few large trees and recently regenerated stems of small diameter, whereas dry sclerophyll sites often contained a range of diameter classes. As a result, woodland sites had a smaller stand basal area and a larger quadratic mean dbh, than dry sclerophyll sites with a comparable density of large trees.

In addressing the four criteria, (number of live stems ha\(^{-1}\) > 40cm dbh):

1. Was normally distributed
2. Did not differentiate between the two communities occurring in the study sites, with similar numbers of large trees in woodland (28.7 stems ha\(^{-1}\)) and dry sclerophyll (35.7 stems ha\(^{-1}\)) sites;
3. Was weakly correlated with stand basal area (r = 0.56) and quadratic mean dbh (r = 0.38);
4. Was straightforward to measure because 75% of sites contained less than 6 stems of this size in an 0.1ha plot.

6.3.10 Coarse woody debris: total log length and total large log length

Coarse woody debris was defined as woody debris with a small end diameter > 10cm and a length > 0.5m (Woldendorp et al., 2004). It was quantified in terms of log cover (%); total log volume (m\(^3\)ha\(^{-1}\)), based on the sum of individual log volumes, where each log volume was computed as length x average sectional...
area; a diversity index based on the distribution of coarse woody debris volume across three decay classes; total log length (mha$^{-1}$); and length of large logs (mha$^{-1}$) with a mean diameter greater than 30cm. The threshold for large logs reflected field observations that logs of this size were more likely to provide important ecological functions such as hollows for nesting or the maintenance of damp micro-sites suitable for fungi and invertebrates (MacNally et al., 2001; Dickman, 1991; Smith et al., 1989).

Log cover did not appear to be an appropriate method for quantifying coarse woody debris because of the low levels of log cover present in dry sclerophyll and woodland communities. This was reflected in the distribution of log cover across study sites, which was highly skewed with more than 25% of sites recording zero log cover compared to less than 2.5% sites recording zero log volume. The mean log cover was 1.5% and the maximum cover recorded was 6.9%. At these low levels of cover, the use of line intersect estimates based on 10 x 1m lengths, may have been an insufficient sampling intensity to provide accurate estimates of log cover. The use of cover classes as advocated by Watson et al. (2001) was also considered inappropriate because all sites fell within their lowest cover class of 0-10%.

In contrast to log cover, log volume was approximately normally distributed and ranged from a minimum of 0 to a maximum of 51.8 m$^3$ha$^{-1}$, with a mean of 15.4 m$^3$ha$^{-1}$. There was no significant difference between average log volumes recorded in woodland and dry sclerophyll sites, although the trend was for woodland sites to have lower volumes (11.1 m$^3$ha$^{-1}$) than dry sclerophyll sites (17.6 m$^3$ha$^{-1}$). When stratified by decay class, most volume occurred in decay class three (gives when kicked), which had a mean of 9.2 m$^3$ha$^{-1}$, followed by decay class two (sound when kicked, and contains cracks, hollows or pipes), with a mean of 5.1 m$^3$ha$^{-1}$, and decay class one (sound when kicked, and does not contain cracks, hollows or pipes), with a mean of 1.1 m$^3$ha$^{-1}$. There were no significant differences between woodland and dry sclerophyll sites in average decay class volumes.

The diversity of coarse woody debris was considered to be a measure of the
relative abundance of coarse woody debris occurring in different decay classes. This reflects the use by fauna of a variety of log conditions, ranging from dry intact hollow logs for nesting and refuge (Woinarski et al., 1997; Smith et al., 1989) to decomposing moist logs which provide a substrate for food resources such as hypogaeal fungi and invertebrates (Woodgate et al., 1994; Dickman, 1991). Coarse woody debris diversity was quantified by constructing an index, which scored sites according to the log volume occurring in each decay class. To do this, volume classes were defined for each decay class based on the quartiles of the distribution of volume for that decay class. These volume classes were then used to award a score from 0-3 for each decay class (Table 20). The Coarse Woody Debris Diversity Index (CWDI) score was the sum of the scores for the three decay classes. This gave a maximum score of 9, for sites with volumes exceeding the 75th percentile in all decay classes, and a minimum score of 0, for sites with volumes less than the 25th percentile in all decay classes.

Table 20. Scoring an index of coarse woody debris diversity on the basis of decay class volume. For each decay class a score from 0-3 was awarded by comparing log volume to the quartiles of the distribution of volume for that decay class. The maximum score on the index was 9 and the minimum 0.

<table>
<thead>
<tr>
<th>Quartile</th>
<th>Decay class 1 volume (m³ha⁻¹)</th>
<th>Decay class 2 volume (m³ha⁻¹)</th>
<th>Decay class 3 volume (m³ha⁻¹)</th>
<th>Score</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-25%</td>
<td>0-0.06</td>
<td>0-1.84</td>
<td>0-2.94</td>
<td>0</td>
</tr>
<tr>
<td>25-50%</td>
<td>0.06-0.48</td>
<td>1.84-4.01</td>
<td>2.94-7.11</td>
<td>1</td>
</tr>
<tr>
<td>50-75%</td>
<td>0.48-1.68</td>
<td>4.01-7.97</td>
<td>7.11-13.87</td>
<td>2</td>
</tr>
<tr>
<td>75-100%</td>
<td>&gt;1.68</td>
<td>&gt;7.97</td>
<td>&gt;13.87</td>
<td>3</td>
</tr>
</tbody>
</table>

The CWDI was normally distributed, and differentiated more effectively between study sites than log volume, with significantly higher coarse woody debris diversity in dry sclerophyll sites (5.0) than in woodland sites (3.4). However, the CWDI was not adopted as a core attribute because it was time consuming to measure in the field, requiring data describing log length, log average diameter, and log decay class. It was also highly correlated with the (total log length ha⁻¹) (r = 0.83), which better met the criteria for selection as a core attribute. This
relationship also indicated that the diversity of woody debris increased more slowly than the total length of logs, so that for example a four fold increase in total log length would on average only produce a two fold increase in coarse woody debris diversity.

In addressing the four criteria for a core attribute, (total log length ha$^{-1}$):

1. Was normally distributed and ranged from a minimum of 0 to a maximum of 1530 mha$^{-1}$, with a mean of 458 mha$^{-1}$;
2. Differentiated between woodland and dry sclerophyll sites, with significantly more total log length in dry sclerophyll sites (519 mha$^{-1}$) than in woodland sites (182 mha$^{-1}$);
3. Was correlated with log volume ($r = 0.85$), CWDI ($r = 0.83$), stand basal area of live trees ($r = 0.67$) and canopy cover ($r = 0.62$);
4. Was straightforward to measure because individual log lengths could be efficiently estimated to the nearest 0.5m by eye or by pacing.

The variable (total log length ha$^{-1}$) was relatively weakly correlated with (large log length ha$^{-1}$) ($r = 0.57$) indicating it was a poor surrogate for large logs, which was identified as one of four key structural attributes distinguishing oldgrowth temperate forests from earlier successional stages (e.g. Woodgate et al., 1994; Franklin et al., 1981). The variable (total large log length ha$^{-1}$) was therefore also included as a core attribute.

In addressing the four criteria, (total large log length ha$^{-1}$):

1. Was normally distributed, and ranged from a minimum of 0 to a maximum of 193 mha$^{-1}$ with a mean of 35 mha$^{-1}$;
2. Did not differentiate between woodland and dry sclerophyll communities, with no significant differences in total large log length between woodland (34.5 mha$^{-1}$) and dry sclerophyll sites (35.7 mha$^{-1}$), although this may reflect increased numbers of large logs in woodland sites due to the past practice of culling overstorey trees;
3. Was correlated with log volume ($r = 0.83$) and CWDI ($r = 0.69$);
4. Was easy to measure because on average only 10% of logs were large.
6.3.11 Litter dry weight

Litter was defined as all dead organic matter <10cm in diameter, covering but not yet incorporated into the uppermost soil horizon. Litter was quantified in terms of horizontal cover (%) and dry weight (tha⁻¹). Litter cover was measured to the nearest 1% using 10 x 1m line intersect estimates, and was modelled as a continuous variable. Litter cover was normally distributed and ranged from a minimum of 30% to a maximum of 91%, with a mean of 71%. Fifty-eight percent of sites exceeded 70% cover, although there was significantly more litter cover in dry sclerophyll sites (78%) than in woodland sites (59%). These high levels of litter cover meant that the use of cover classes as suggested by Watson et al. (2001) or by Newsome and Catling (1979) was inappropriate (Table 21).

Table 21. Percentage of 48 study sites (dry sclerophyll and woodland sites combined) occurring in each of the four litter cover classes suggested by Newsome and Catling (1979), and Watson et al. (2001), as a component of Habitat Complexity Score.
* Indicates a cover class that is redundant.

<table>
<thead>
<tr>
<th>Cover class</th>
<th>Percentage of sites</th>
<th>Cover class</th>
<th>Percentage of sites</th>
</tr>
</thead>
<tbody>
<tr>
<td>0*</td>
<td>0%</td>
<td>0-10*</td>
<td>0%</td>
</tr>
<tr>
<td>0-30*</td>
<td>2%</td>
<td>10-40</td>
<td>4%</td>
</tr>
<tr>
<td>30-70</td>
<td>40%</td>
<td>40-70</td>
<td>38%</td>
</tr>
<tr>
<td>&gt;70</td>
<td>58%</td>
<td>&gt;70</td>
<td>58%</td>
</tr>
</tbody>
</table>

In these dry ecosystems it appeared that the absence of recent fire and slow decomposition rates, maintained high levels of litter cover, so that litter was better described in terms of biomass, which was a function of litter cover and depth, rather than by cover alone (Figure 8). Slow decomposition rates might also account for the relatively weak correlation between vegetation cover <0.5m and either litter cover (r = -0.49), or litter dry weight (r = -0.46).
Figure 8. Litter dry weight distinguished between sites with similarly high levels of litter cover, as indicated by the relationship between litter cover (%) and litter dry weight (tha$^{-1}$) based on data from 16 woodland and 32 dry sclerophyll sites.

Litter cover = 88.6 – 151.5 (Litter dry weight)$^{-1}$  \( r^2 = 0.67, p < 0.0001 \)
Litter biomass was quantified in terms of litter dry weight (tha\(^{-1}\)), based on five samples taken at points systematically located within a 50mx20m quadrat. There was a good fit between litter cover and the negative reciprocal of litter dry weight \((r^2 = 0.67)\), confirming that litter dry weight distinguished between sites with similarly high levels of litter cover (Figure 8). Litter dry weight also met all four criteria for selection as a core attribute; it:

1. Was normally distributed amongst study sites and ranged from a minimum of 3.2 tha\(^{-1}\) to a maximum of 22.0 tha\(^{-1}\), with a mean of 11.8 tha\(^{-1}\);
2. Distinguished between woodland and dry sclerophyll communities, with significantly higher levels of litter dry weight in dry sclerophyll sites (14.3 tha\(^{-1}\)), compared to woodland sites (6.8 tha\(^{-1}\));
3. Was correlated with litter cover \((r = 0.73)\), stand basal area of live trees \((r = 0.61)\) and canopy projective cover \((r = 0.62)\);
4. Was straightforward to collect in the field, although there was a time delay of 2-4 days associated with oven drying litter.

**6.4 Testing for redundancy in the 13 core attributes**

In the third and final stage of the analysis, Principal Components Analysis was used to test for redundancy amongst the core attributes identified in stage one. This was done using Gabriel plots (Legendre and Legendre, 1998) to represent core attributes as vectors showing the relative importance of each attribute in explaining variation between sites, and the degree of correlation between core attributes.

**6.4.1 Principal Components Analysis**

The 13 core attributes identified in the previous sections located the 48 study sites in a 13 dimensional ecological space. It was impossible to visualise this number of dimensions directly, or to judge the relative importance of each attribute in determining the position of study sites in this ecological space. To address these issues Principal Components Analysis was used to reduce the number of dimensions by reconstructing the ecological space in terms of new axes or principal components. In this approach each principal component is a linear combination of the variables that defined the original axes. The first
principal component is constructed so that it captures the maximum amount of variation in the data points that occur in the ecological space. In this study the first principal component is that linear combination of the 13 core attributes which captures most variation in the position of the 48 study sites in the original 13 dimensional space. The second principal component is constructed orthogonal to the first and is oriented so that it captures the maximum amount of remaining variation. Successive principal components can be constructed in this way until there are as many principal components as original axes at which point all of the variation in the data points will have been captured. Each principal component is always a linear combination of the original variables with successive principal components explaining progressively less variation in the data points (Gauch, 1982).

6.4.2 Gabriel plots

If the first two principal components capture a substantial amount of the total variation in the data points then these components can be used to map, or ordinate, the data points in two dimensions and so investigate any inherent structure in the data. The original axes or variables can also be represented as vectors in this two dimensional space using a graphical technique pioneered by Gabriel (1971). The length of these vectors shows the relative importance of each variable in explaining variation between data points, and the angle between vectors the degree of correlation between variables. The least correlated arrangement for two vectors is an orientation of 90 degrees to each other. For three or more vectors the least correlated arrangement is when the angle between adjacent vectors is maximised, so that vectors radiate equally in a pattern similar to the spokes of a wheel (Figure 9). This graphical technique was used to investigate redundancy amongst the 13 core attributes. The optimum arrangement for the core attributes was assumed to be one in which the attributes were of equal importance in explaining variation between sites, and any correlations between core attributes were minimised. This would correspond to the 13 vectors having similar length and being oriented in a radial pattern similar to Figure 9B.
Figure 9. Using Principal Components Analysis the original axes or variables in a multidimensional space can be represented as vectors in a two dimensional space defined by the first two principal components (PC1 and PC2). These vectors show the relative importance of each variable in explaining variation between data points in this reduced space (vector length), and the degree of correlation between the different variables (angle between vectors) (A). The least correlated arrangement is when the angle between adjacent vectors is maximised, so that vectors radiate equally in a pattern similar to the spokes of a wheel (B).

Principal Components Analysis was performed using the JMP statistical software package (SAS Institute, 2001). For this analysis the 13 attributes, which were normally distributed, were standardised to a mean of zero and a standard deviation of 1, to remove any bias due to the different scales of measurements used for each attribute. The first two principal components explained a total of 59% of the variation between study sites. The vectors, representing the size and direction of the 13 attributes in the ecological space defined by the first two principal components, are shown in Figure 10.
Chapter 6: Arriving at a core set of attributes

Figure 10. The first two principal components (PC1 and PC2) explained a total of 59% of the variation between study sites in the ecological space mapped by the 13 core attributes. The vectors indicate the magnitude and direction in which the core attributes operate in the reduced space defined by the first two principal components. The vectors correspond to the following core attributes: 1. Vegetation cover <0.5m; 2. dbh<sub>Q</sub>; 3. (total large log length ha<sup>-1</sup>); 4. (number of live stems ha<sup>-1</sup> > 40cm dbh); 5. (total log length ha<sup>-1</sup>); 6. Stand basal area of live trees; 7. ln(number of hollow-bearing trees ha<sup>-1</sup>+1); 8. Litter dry weight; 9. ln(number of dead trees ha<sup>-1</sup>+1); 10. Lifeform richness; 11. Perennial species richness; 12. ln(number of regenerating stems ha<sup>-1</sup>+1); 13. Vegetation cover 0.5-6.0m.

With the exception of ln(number of regenerating stems ha<sup>-1</sup>+1), the vectors in Figure 10 were of comparable length, confirming that 12 of the attributes have a similar weighting in mapping the location of study sites in this two dimensional space. In terms of vector orientation, Figure 10 highlighted six distinct attribute groupings, arranged in a radial pattern that minimised correlations between groups:
1. Perennial species richness, lifeform richness, vegetation cover <0.5m;
2. ln(number of regenerating stems ha\(^{-1}\)), vegetation cover >0.5m, dbh\(Q\);
3. ln(number of dead trees ha\(^{-1}\)+1), litter dry weight;
4. \(\sqrt{\text{total large log length}}\), \(\sqrt{\text{number of live stems ha}^{-1}>40\text{cm dbh}}\);
5. ln(number of hollow-bearing trees ha\(^{-1}\)+1);
6. \(\sqrt{\text{total log length ha}^{-1}}\), stand basal area of live trees;

While attributes within groups were correlated to some degree these relationships were not considered sufficiently robust to justify reducing the number of core attributes. There were three reasons for this decision. First, correlations between attributes within a group did not always operate across both dry sclerophyll and woodland communities; in particular perennial species richness was strongly correlated with lifeform richness in woodland sites (r = 0.84) but not in dry sclerophyll sites (r = 0.38); dbh\(Q\) was correlated with vegetation cover 0.5-6.0m in woodland sites (r = -0.53) but not in dry sclerophyll sites (r = -0.23); ln(number of regenerating stems ha\(^{-1}\)+1) was correlated with vegetation cover 0.5-6.0m in woodland sites (r = 0.63) but not in dry sclerophyll sites (r = -0.14); ln(number of dead trees ha\(^{-1}\)+1) was correlated with litter dry weight in woodland sites (r = 0.54) but not in dry sclerophyll sites (r = 0.06); \(\sqrt{\text{total large log length}}\) was correlated with (number of live stems ha\(^{-1}\)>40cm dbh) in dry sclerophyll sites (r = 0.68) but not in woodland sites (r = 0.16).

Second, correlations between core attributes did not guarantee a similar association with other related attributes. For example, stand basal area of live trees functioned as a surrogate for canopy cover (r = 0.77) while \(\sqrt{\text{total log length ha}^{-1}}\) did not (r = 0.48), even though it was correlated with stand basal area of live trees (r = 0.67); similarly while \(\sqrt{\text{total log length ha}^{-1}}\) was a surrogate for coarse woody debris diversity (r = 0.83), basal area of live trees was not (r = 0.54).

Third, although attributes within groups were correlated to some degree, they often quantified distinctly different structural elements. For example lifeform richness and perennial species richness described compositional diversity, whereas vegetation cover <0.5m described the physical arrangement of ground
foliage; similarly ln(number of regenerating stems ha^{-1}+1) described the key functional process of overstorey regeneration, whereas vegetation cover 0.5-6.0m described the physical arrangement of foliage in the shrub layer.

6.5 Conclusions

The three-stage analysis presented in this chapter demonstrated that data from 48 sites representing a broad suite of condition states in woodland and dry sclerophyll forest could be used to identify core attributes for characterising structural complexity in these communities. The data also indicated that these core attributes could be quantified as continuous variables, which when suitably transformed, had an approximately normal distribution across study sites. This approach maximised the ability of core attributes to distinguish between study sites, and demonstrated that the use of classes to measure attributes was inappropriate because it lost information and masked differences between sites.

The data also indicated the scale at which different attributes operated, and hence how they should be scored as part of an index of structural complexity. For example total log length was quantified in terms of √(total log length) because this variable was normally distributed and well correlated with coarse woody debris diversity. Hence a four-fold increase in log length should double rather than quadruple the score for this attribute in an index of structural complexity.

Although Principal Components Analysis highlighted six groupings within which attributes were correlated to some degree, these relationships were not considered sufficiently robust to justify reducing the number of core attributes. The 13 core attributes were therefore accepted as an optimum set of structural attributes, in which no single attribute appeared to be redundant. In the next chapter, I describe how these 13 attributes were combined in an index of structural complexity.
Chapter 7: An index for rapidly assessing stand structural complexity in dry sclerophyll forests and woodlands of south-eastern Australia

Chapter summary

The thirteen key attributes identified in Chapter 6 are combined in an index of structural complexity. In this index attributes are rescaled as a score from 0-10, using equations that model attribute score as a function of the raw attribute data. Separate sets of equations were derived for woodland and dry sclerophyll sites, because the two communities were considered structurally distinct from each other. The rescaled attributes were combined in a simple additive index, in which the 13 attributes carried equal weight, each accounting for 10 points in a total index value of 130. Sensitivity analysis indicated that the performance of this index was not sensitive to the weighting of attributes. For 10,000 simulations there was a high correlation between scores on the unweighted index and scores on a randomly weighted index. The unweighted index was therefore accepted as the most suitable format for the final index. The chapter concludes by demonstrating the practical application of the proposed index in a user-friendly spreadsheet, designed to allow landowners and managers to assess the condition of their vegetation, and to identify management options.

7.1 Introduction

In this chapter I address the fifth and final part of the methodology for developing an index of structural complexity, which was proposed in Chapter 3. This involved combining the thirteen core attributes identified in the previous chapter, in a simple index. The purpose of this index is to act as a summary variable for the thirteen core attributes, and thereby provide a quantitative tool for ranking stands in terms of their structural complexity, and by association, their biodiversity and vegetation condition. In the first stage of designing the index, a prototype index is constructed in which all attributes carry equal weight. This is achieved by rescaling attributes as a score from 0-10, so that any weighting due to the different measurement units used for the various attributes is removed. In the second stage attributes are randomly weighted and sensitivity analysis used to determine the most appropriate weightings to use in the index. To conclude the completed index is incorporated into a user-friendly
Chapter 7: An index for rapidly assessing stand structural complexity

spreadsheet for use by landowners and managers.

7.2 Combining attributes in an index

7.2.1 Index framework

A simple mathematical framework was used to construct the index of stand structural complexity. In this framework the thirteen core attributes identified previously were added together to give a total score or index value. At any site the score for a particular attribute reflected the abundance of that attribute, so that up to a threshold level, sites with more of a particular attribute scored higher for this attribute. This scoring approach reflected established relationships between fauna habitat and the abundance of attributes, and acknowledged that excessive levels of an attribute were unlikely to contribute additional habitat when that attribute is no longer a limiting resource.

In particular, the review presented in Chapter 4 indicated a typically positive association between a given faunal group and the attributes that characterised its habitat requirements. Where a negative association was reported this was because the attribute in question was negatively correlated with a critical component of habitat for that faunal group. For example, Catling et al. (2000) reported a negative association between the presence of the eastern grey kangaroo and the cover of shrubs. This reflected the preference of these animals for an open understorey with a ground cover of grasses, an attribute which is negatively correlated with shrub cover (Seddon et al., 2001). This example also demonstrates that interactions between attributes usually preclude very high levels in all attributes. I treated the structural complexity of a stand as a measure of how these interactions have distributed site resources amongst different structural attributes. I took the range of stand structures present in the environment to reflect different distributions of these resources amongst structural attributes; where simple structures often have high levels in a few attributes, with many attributes absent or present at very low levels, and complex structures often have medium to high levels in a wide range of attributes. This was analogous to the many approaches which have quantified diversity in terms of the abundance and richness of elements (Magurran, 1988).

The decision to use a simple additive index to summarise the attribute
interactions underpinning structural complexity reflected the results of the review of indices presented in Chapter 3. These highlighted the advantages of a simple framework, such as the ease of incorporating multiple attributes and interpreting the index in terms of real stand conditions.

### 7.2.2 Rescaling attributes

To provide an objective starting point for applying this index framework, attributes were rescaled as a score from 0-10, using equations that modelled attribute score as a function of the raw attribute data. This approach was possible because each of the thirteen core attributes had been measured as a continuous variable, and had an approximately normal distribution across the study sites. Separate sets of equations were derived for woodland and dry sclerophyll sites, because the two communities were considered structurally distinct from each other. For example, 11 of the 13 core attributes had significantly (p<0.05) different levels in woodland compared to dry sclerophyll forest (Table 17, Chapter 6). A woodland site was therefore scored relative to the range of observed attribute levels in woodland sites, and a dry sclerophyll site was scored relative to the range of observed attribute levels in dry sclerophyll sites. This was in accordance with the review presented in Chapter 3, which concluded that structural complexity is a relative rather than absolute concept, and that structural attributes should be scored relative to the range of values occurring in stands of a comparable vegetation community.

For each attribute, two equations were derived, using regression analysis to fit a linear relationship between attribute score and the quartile midpoints of the attribute distributions in woodland and dry sclerophyll sites. The quartile midpoints corresponded to the 12.5, 37.5, 62.5 and 87.5 percentiles of the attribute distributions and were awarded scores of 2.5, 5, 7.5 and 10 respectively (Table 21 and Figure 11). The attribute level corresponding to the maximum score of 10 was set by the 87.5 percentile to prevent extreme values distorting the scaling of scores. The set of equations for scoring the thirteen attributes is shown in Table 22. These equations were constrained so that the rescaled score was always between 0 and 10, and a zero value in the raw data always resulted in a zero score in the rescaled data. Using this approach very little information was lost in the rescaling process, and the rescaled attribute scores were highly correlated with the original raw data (Table 23).
Table 21: Quartile midpoints were used to rescale attribute data as a score from 0-10. The quartile midpoints corresponded to the 12.5, 37.5, 62.5 and 87.5 percentiles of the attribute distribution and were awarded a score of 2.5, 5, 7.5 and 10 respectively. Woodland and dry sclerophyll forest sites were treated as structurally distinct communities, and were scored relative to the range of observed attribute levels in each community; (a) Woodland sites (b) Dry sclerophyll sites. * Quartile mid-points are in the original, as opposed to transformed measurement units.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Percentile (Score)</th>
<th>*Quartile mid-points</th>
<th>12.5</th>
<th>37.5</th>
<th>62.5</th>
<th>87.5</th>
</tr>
</thead>
<tbody>
<tr>
<td>Perennial species richness (per 400m²)</td>
<td>(a) 6.8</td>
<td>(b) 12.6</td>
<td>10.4</td>
<td>14.0</td>
<td>21.7</td>
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<tr>
<td>Lifeform richness (per 400m²)</td>
<td>(a) 3.9</td>
<td>(b) 6.9</td>
<td>5.2</td>
<td>7.0</td>
<td>8.8</td>
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<tr>
<td>Stand basal area of live trees (m²ha⁻¹)</td>
<td>(a) 9.2</td>
<td>(b) 15.5</td>
<td>12.2</td>
<td>17.1</td>
<td>33.7</td>
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<tr>
<td>Quadratic mean diameter of live stems (cm)</td>
<td>(a) 21.8</td>
<td>(b) 16.5</td>
<td>31.0</td>
<td>42.3</td>
<td>62.3</td>
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<tr>
<td>Vegetation cover &lt;0.5m (%)</td>
<td>(a) 27.5</td>
<td>(b) 9.9</td>
<td>46.2</td>
<td>62.7</td>
<td>80.3</td>
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<tr>
<td>Vegetation cover 0.5-6.0m (%)</td>
<td>(a) 0.5</td>
<td>(b) 1.3</td>
<td>1.4</td>
<td>2.7</td>
<td>4.6</td>
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<td>In(number of regenerating stems ha⁻¹+1)</td>
<td>(a) 24</td>
<td>(b) 66</td>
<td>76</td>
<td>262</td>
<td>833</td>
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<tr>
<td>In(number of hollow bearing trees ha⁻¹+1)</td>
<td>(a) 2.1</td>
<td>(b) 3.4</td>
<td>5.8</td>
<td>9.0</td>
<td>21.6</td>
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<tr>
<td>(number of live stems ha⁻¹ &gt; 40cm dbh)</td>
<td>(a) 10.1</td>
<td>(b) 4.4</td>
<td>19.3</td>
<td>33.5</td>
<td>66.1</td>
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<tr>
<td>In(number of dead trees ha⁻¹+1);</td>
<td>(a) 1.2</td>
<td>(b) 5.3</td>
<td>3.9</td>
<td>14.4</td>
<td>37.7</td>
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<tr>
<td>(total log length) (mha⁻¹)</td>
<td>(a) 26.5</td>
<td>(b) 86.5</td>
<td>128.9</td>
<td>267.3</td>
<td>458.0</td>
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<tr>
<td>(total large log length) (mha⁻¹)</td>
<td>(a) 14.1</td>
<td>(b) 14.1</td>
<td>34.6</td>
<td>72.9</td>
<td>98.4</td>
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<td>Litter dry weight (tha⁻¹)</td>
<td>(a) 3.4</td>
<td>(b) 8.9</td>
<td>4.5</td>
<td>7.3</td>
<td>13.0</td>
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Figure 11. Attributes were rescaled as a score from 0-10 by fitting linear regressions to the quartile midpoints of the attribute distributions in woodland and dry sclerophyll sites. The quartile midpoints corresponded to the 12.5, 37.5, 62.5 and 87.5 percentiles of the attribute distributions and were awarded scores of 2.5, 5, 7.5 and 10 respectively. The process is illustrated here for the attribute $\sqrt{\text{total log length mha}^{-1}}$. 
Table 22: Regression equations for scoring attributes on a scale of 0-10 as a function of the raw attribute data. Regression equations were fitted to the 12.5, 37.5, 62.5, and 87.5 percentiles of each attribute distribution, which had been awarded a score of 2.5, 5, 7.5, and 10 respectively. The equations were constrained so that the attribute score was always between 0 and 10.

(a) Woodland sites (b) Dry sclerophyll sites.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Regression equation</th>
<th>$r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Perennial species richness (per 400m²)</td>
<td>(a) $\text{Score} = -0.315 + 0.497(\text{Perennial species richness})$</td>
<td>0.96</td>
</tr>
<tr>
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<td>(b) $\text{Score} = -5.022 + 0.571(\text{Perennial species richness})$</td>
<td>0.98</td>
</tr>
<tr>
<td>Lifeform richness (per 400m²)</td>
<td>(a) $\text{Score} = -3.114 + 1.508(\text{Lifeform richness})$</td>
<td>0.99</td>
</tr>
<tr>
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<td>(b) $\text{Score} = -20.546 + 3.291(\text{Lifeform richness})$</td>
<td>0.96</td>
</tr>
<tr>
<td>Stand basal area (m²ha⁻¹)</td>
<td>(a) $\text{Score} = 0.610 + 0.305(\text{Stand basal area})$</td>
<td>0.92</td>
</tr>
<tr>
<td></td>
<td>(b) $\text{Score} = -3.269 + 0.356(\text{Stand basal area})$</td>
<td>0.98</td>
</tr>
<tr>
<td>Quadratic mean dbh (cm)</td>
<td>(a) $\text{Score} = -0.902 + 0.182(\text{Quadratic mean dbh})$</td>
<td>0.98</td>
</tr>
<tr>
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<td>(b) $\text{Score} = -4.454 + 0.426(\text{Quadratic mean dbh})$</td>
<td>0.99</td>
</tr>
<tr>
<td>Vegetation cover &lt;0.5m (%)</td>
<td>(a) $\text{Score} = -0.468 + 0.126(\text{Vegetation cover &lt;0.5m})$</td>
<td>0.99</td>
</tr>
<tr>
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<td>(b) $\text{Score} = 0.331 + 0.200(\text{Vegetation cover &lt;0.5m})$</td>
<td>0.99</td>
</tr>
<tr>
<td>Vegetation cover 0.5-6m (%)</td>
<td>(a) $\text{Score} = 1.225 + 2.055(\text{Vegetation cover 0.5-6.0m})$</td>
<td>0.95</td>
</tr>
<tr>
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<td>(b) $\text{Score} = 0.624 + 1.208(\text{Vegetation cover 0.5-6.0m})$</td>
<td>0.98</td>
</tr>
<tr>
<td>In(number of regenerating stems ha⁻¹+1)</td>
<td>(a) $\text{Score} = -4.147 + 2.100ln(\text{number of regenerating stems ha⁻¹+1})$</td>
<td>0.99</td>
</tr>
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<td>(b) $\text{Score} = -11.89 + 3.477ln(\text{number of regenerating stems ha⁻¹+1})$</td>
<td>0.99</td>
</tr>
<tr>
<td>In(number of hollow bearing trees ha⁻¹+1)</td>
<td>(a) $\text{Score} = -0.065 + 3.253ln(\text{number of hollow bearing trees ha⁻¹+1})$</td>
<td>0.99</td>
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<td>(b) $\text{Score} = -0.190 + 2.012ln(\text{number of hollow bearing trees ha⁻¹+1})$</td>
<td>0.98</td>
</tr>
<tr>
<td>(number of live stems ha⁻¹ &gt; 40cm dbh)</td>
<td>(a) $\text{Score} = -0.536 + 1.287(\text{number of live stems ha⁻¹ &gt; 40cm dbh})$</td>
<td>0.97</td>
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<td>(b) $\text{Score} = -0.038 + 1.028(\text{number of live stems ha⁻¹ &gt; 40cm dbh})$</td>
<td>0.99</td>
</tr>
<tr>
<td>In(number of dead trees ha⁻¹+1):</td>
<td>(a) $\text{Score} = 1.119 + 2.472ln(\text{number of dead trees ha⁻¹+1})$</td>
<td>0.96</td>
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<td>(b) $\text{Score} = -0.404 + 1.767ln(\text{number of dead trees ha⁻¹+1})$</td>
<td>0.96</td>
</tr>
<tr>
<td>(total log length) (mha⁻¹)</td>
<td>(a) $\text{Score} = -0.016 + 0.462 (\text{total log length ha⁻¹})$</td>
<td>0.99</td>
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<td>(b) $\text{Score} = -0.276 + 0.297 (\text{total log length ha⁻¹})$</td>
<td>0.98</td>
</tr>
<tr>
<td>(total large log length) (mha⁻¹)</td>
<td>(a) $\text{Score} = -0.568 + 0.991 (\text{total large log length ha⁻¹})$</td>
<td>0.99</td>
</tr>
<tr>
<td></td>
<td>(b) $\text{Score} = 0.197 + 0.750 (\text{total large log length ha⁻¹})$</td>
<td>0.96</td>
</tr>
<tr>
<td>Litter dry weight (tha⁻¹)</td>
<td>(a) $\text{Score} = 0.556 + 0.788(\text{Litter dry weight})$</td>
<td>0.94</td>
</tr>
<tr>
<td></td>
<td>(b) $\text{Score} = -3.850 + 0.710(\text{Litter dry weight})$</td>
<td>0.99</td>
</tr>
</tbody>
</table>
Table 23: Attribute scores on a scale from 0-10 were highly correlated with the original attribute data, indicating there was little information loss in the rescaling process. All correlations were highly significant (p<0.0001).

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Correlation (Pearson’s r) between raw data and rescaled score</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Woodland</td>
</tr>
<tr>
<td>Perennial species richness (per 400m$^2$)</td>
<td>0.969</td>
</tr>
<tr>
<td>Lifeform richness (per 400m$^2$)</td>
<td>0.997</td>
</tr>
<tr>
<td>Stand basal area of live trees (m$^2$ha$^{-1}$)</td>
<td>0.982</td>
</tr>
<tr>
<td>Quadratic mean diameter of live stems (cm)</td>
<td>0.967</td>
</tr>
<tr>
<td>Vegetation cover &lt;0.5m (%)</td>
<td>0.997</td>
</tr>
<tr>
<td>Vegetation cover 0.5-6.0m (%)</td>
<td>0.981</td>
</tr>
<tr>
<td>In(number of regenerating stems ha$^{-1}+1$)</td>
<td>0.995</td>
</tr>
<tr>
<td>In(number of hollow bearing trees ha$^{-1}+1$)</td>
<td>0.989</td>
</tr>
<tr>
<td>(number of live stems ha$^{-1} &gt;$ 40cm dbh)</td>
<td>0.978</td>
</tr>
<tr>
<td>In(number of dead trees ha$^{-1}+1$)</td>
<td>0.987</td>
</tr>
<tr>
<td>(total log length) (mha$^{-1}$)</td>
<td>0.994</td>
</tr>
<tr>
<td>(total large log length) (mha$^{-1}$)</td>
<td>0.999</td>
</tr>
<tr>
<td>Litter dry weight (tha$^{-1}$)</td>
<td>0.972</td>
</tr>
</tbody>
</table>

### 7.2.2 Prototype Index

The rescaled attributes were combined in a prototype index, in which attributes carried equal weight, and each accounted for 10 points out of a total index value of 130. This index value was then expressed as a percentage. The prototype index differentiated between sites in both woodland and dry sclerophyll forest, and was approximately normally distributed amongst study sites in these communities (Figure 12). The range and mean value of the prototype index were similar for the two communities; in woodland the index ranged from a minimum of 43.2% to a maximum of 78.2% with a mean of 58.5%, and in dry sclerophyll forest the index ranged from a minimum of 34.9% to a maximum of 83.9% with a mean of 61.1%. These results confirmed that the
index was scoring sites relative to the expected levels for attributes in each community, so that although woodland sites had inherently lower levels for most attributes than dry sclerophyll forest (Table 21), high quality woodland sites registered index scores comparable to high quality dry sclerophyll forest sites.

The prototype index appeared to rank sites in a logical way, with least modified stands with oldgrowth characteristics scoring more highly than modified stands with regrowth characteristics (Figure 13). However, the index also indicated that some regrowth stands could achieve moderate to high levels of structural complexity (Figure 13b and 13e). In this study, these were regrowth stands containing legacies from pre-disturbance conditions, such as large mature trees, dead trees and fallen logs. This was an important result because it indicated that there was no single ideal stand structure, but instead a variety of different ways in which attributes could combine to produce a range of structurally complex stands.

The maximum values achieved on the prototype index (woodland 78.2% and dry sclerophyll forest 83.9%) confirmed that interactions between attributes made it unlikely for a site to achieve maximum (100%) levels in all attributes. For example high litter loads were often associated with reduced ground vegetation cover, and a high density of regeneration was often associated with open stands having a low basal area. As a result, high structural complexity was more likely to be associated with intermediate levels in many attributes, rather than high levels in a few attributes.
Figure 12. The prototype (unweighted) index differentiated between study sites, with index values approximately normally distributed in both (a) woodland and (b) dry sclerophyll sites
Figure 13. The prototype index ranked sites in a logical way in woodland (a,b,c) and dry sclerophyll forest (d,e,f). Least modified sites with oldgrowth characteristics (c 78%, f 73%) scored higher than highly modified sites (a 49%, d 39%). Some regrowth stands, which contained legacies from the pre-disturbance conditions (e.g. large living and dead trees), achieved moderate to high scores (b 72%, e 60%).

7.3 Weighting of attributes in the index

The weighting of attributes is a feature of many indices (e.g. Gibbons et al., 2004, Parkes et al. 2003, Van Den Meersschaut and Vandekerkhove, 1998) and is potentially useful if there is a clear rationale for valuing the presence of some attributes more than others, and if weighting can bias the performance of
the index in the desired manner. For example in a management context it could be useful to weight attributes which take a long time to replace, such as hollow bearing trees, above attributes which can respond relatively rapidly to active management, such as vegetation cover.

As a first step in deciding how to weight attributes, I tested whether weighting actually affected the operation of the prototype index. Sensitivity analysis was used to test the null hypothesis that the performance of the prototype index was independent of the weighting of the attributes in the index. To do this a random weight between 0 and 2 was applied to each attribute in the prototype index. These weights were generated from an even distribution so that all weightings occurred with an equal probability (e.g. a 0.01 was as likely as a 0.63 or a 1.47 or a 1.99). With all thirteen attributes randomly weighted, the index was reapplied to the study sites and a new set of index scores produced. The correlation between these scores and the scores using the unweighted index was then quantified using Pearson’s correlation coefficient (Figure 14). This process was repeated 10,000 times, and each time a new and independent set of random weightings was applied to the thirteen attributes.

The results of this analysis indicated that the performance of the index was not sensitive to the weighting of attributes. For the 10,000 simulations there was a high correlation between scores on the randomly weighted index and scores on the unweighted index (Figure 15). The mean correlation was 0.9475, and the distribution of the correlation coefficients generated by the simulations was skewed towards high values, with 97.5% of simulations having a correlation exceeding 0.878. The robust performance of the unweighted index was attributed to the inclusion of thirteen attributes, each of which was normally distributed and able to differentiate between study sites. Consequently any substantial change in the weighting of a particular attribute was buffered by the information contained in the other attributes. The unweighted or prototype index was therefore accepted as the most suitable format for the final index.
Figure 14. One of 10,000 simulations used to quantify the correlation between unweighted and weighted index scores for 48 woodland and dry sclerophyll forest sites. For each simulation thirteen random weights between 0 and 2 were generated from an even distribution and simultaneously applied to each of the thirteen attributes in the prototype index. The correlation between weighted and unweighted index scores for study sites was quantified using Pearson’s correlation coefficient. In this simulation the random weightings were 0.17, 0.02, 0.70, 0.54, 0.90, 1.60, 0.72, 1.27, 1.15, 1.70, 1.67, 0.75, 1.85, and the correlation between unweighted and weighted index scores was 0.942.
Figure 15 Distribution of the correlation (Pearson’s r) between weighted and unweighted index scores in 48 study sites, for 10,000 random weightings of index attributes. The weightings ranged from 0 to 2 and were drawn from an even distribution. The 95% confidence interval for the mean correlation was 0.9475 ± 0.0005
7.4 Applying the index in a user-friendly spreadsheet

To facilitate the use of the index by land managers it was incorporated into a simple spreadsheet. This spreadsheet had four aims:

1. To provide a simple means for entering raw data for each of the thirteen attributes in the index;
2. To use this raw data to calculate an index value;
3. To generate useful outputs to aid interpretation of the index value in terms of real stand conditions and to highlight future management options – for example by informing managers of the levels at which attributes are present, and where necessary suggesting management actions to improve these levels;
4. To provide an option for calculating an index value using a subset of the thirteen attributes when data for all attributes was unavailable, and to use sensitivity analysis to estimate the uncertainty associated with this value.

The spreadsheet was built using Microsoft EXCEL software (Microsoft Corporation, 2001). To maintain a simple interface all programming was hidden from the user, however it has been left unhidden for inspection on the CD ROM accompanying this thesis. The operation of the spreadsheet is summarised in Figure 16, and described in detail in the following sections.

7.4.1 Data entry

Stand level means for each of the thirteen attributes in the index were entered on the first page of the spreadsheet (Figure 17). It was assumed that these means would be the average of three representative samples quantified using the plot based measurement system described in Chapter 5. To make data entry as simple as possible, the spreadsheet was designed to accept means in their original measurement units, rather than as the transformed variables used in the index. For example, the number of hollow bearing trees was entered as a count of trees ha\(^{-1}\). The spreadsheet then transformed this raw count to the variable \(\ln(\text{hollow bearing trees ha}^{-1}+1)\).

To complete the data entry stage the vegetation community, woodland or dry sclerophyll forest, was specified (Figure 17). This decision was used as a switch to select the appropriate set of equations (Table 22), programmed to automatically score attributes using the raw data that had been entered.
Figure 16. Summary of data requirements, and outputs produced, by a user-friendly spreadsheet incorporating the final version of the index.

DATA ENTRY
Enter site level data for each of 13 attributes.
Choose community (woodland or DSF)
Appropriate equations automatically calculate attribute scores

INDEX USING A SUBSET OF ATTRIBUTES
- Index value calculated using a subset of the thirteen attributes
- Sensitivity analysis estimates uncertainty in the performance of the index

INDEX USING THIRTEEN ATTRIBUTES
- Index value calculated using all thirteen attributes
- Index value used to rank sites relative to reference (study) sites

OUTPUTS FOR INDIVIDUAL ATTRIBUTES
- Status of individual attributes assessed – high, average or low
- Management actions recommended
Figure 17. Data entry page of the index spreadsheet. Site level means are entered for each attribute, and the vegetation type specified by entering either w (woodland) or d (dry sclerophyll forest). Attributes are scored automatically using the appropriate equations for the specified vegetation type. An Index score (%) is calculated and the site ranked relative to a set of reference sites (16 woodland, 32 dry sclerophyll forest).
7.4.2 Calculating an index score and site ranking

An Index score (%) was calculated automatically from the attribute scores, and was used to rank the site relative to scores in a set of reference sites (Figure 17). The reference sites were the 16 woodland and 32 dry sclerophyll forest study sites, which provided a representative sample of the range of condition in the study area (Chapter 5). The ranking therefore added a regional context to the assessment of the quality of the site. It is also a potentially useful tool for identifying sites of high conservation value or for identifying management actions that have maintained site quality.

7.4.3 Calculating an index score using a subset of attributes

The spreadsheet provided the option of calculating an index score using a subset of attributes. This allows the index to be applied to existing data sets, which may not include data for all 13 attributes. It also allows managers to rationalise the effort they put into data collection, because some attributes are more time consuming to quantify than others (eg. litter dry weight compared to lifeform richness).

To apply the index with a subset of attributes, those attributes for which data is missing are left blank. These appear as red cells in the spreadsheet to alert the user (Figure 18). The spreadsheet scores those attributes for which data is entered and calculates an index value (%) using this subset of attributes. To provide a measure of the uncertainty associated with the simplified index value the spreadsheet automatically conducts a sensitivity analysis. To do this a new index value for each of the reference sites is calculated using the specified subset of attributes. The change in rank for each reference site – full index ranking compared to simplified index ranking – is also calculated. The median (50\(^{th}\) percentile) change in rank for sites in the relevant vegetation type is then converted to a percentage. This provides an estimate of the 50% confidence range for the ranking of sites using the simplified index. Similarly the 90\(^{th}\) percentile change in rank for sites, provides an estimate of the 90% confidence range for the ranking of sites (Figure 18).
Figure 18. The spreadsheet provided the option of calculating an index score using a subset of attributes. This simplified index value was calculated by excluding attributes for which data was missing (red cells). Sensitivity analysis provided estimates of the uncertainty in ranking sites using the simplified index value (50% and 90% confidence ranges). The strength of the linear relationship between the ranking of sites using the full index and simplified index was also quantified (r and $r^2$).
As a further guide, the strength of the linear relationship was quantified between the ranking of reference sites using the full index and the rankings using the simplified index. This was reported in terms of Pearson’s correlation coefficient (r), and Rsquared ($r^2$) (Figure 18).

### 7.4.4 Outputs for individual attributes

The second page of the spreadsheet provided information on the status of individual attributes (Figure 19). On this page the level for each attribute was compared to the range of values that occurred in the reference sites. Attribute levels were classed as, high (in the top third of reference site values), average (in the middle third of reference site values) and low (in the bottom third of reference site values). The use of these three classes was considered informative, because it provided a simple scale for highlighting the current status of each attribute and for setting future management options. For example the spreadsheet could be programmed to suggest management actions depending on whether the level of an attribute was low, average or high (Figure 19).

In the next chapter I compare the performance of the index developed above to other indices currently used to quantify fauna habitat or vegetation condition in temperate Australian forest ecosystems.
Figure 19. The second page of the spreadsheet provided information on the status of individual attributes. Attribute levels were classed as high (in the top third of reference site values), average (in the middle third of reference site values) and low (in the bottom third of reference site values). These classes provided a basis for recommending management options.
Chapter 8: Comparing the performance of the index to other indices used in temperate Australian ecosystems

Chapter summary

I compared the performance of my proposed index with five prominent indices used to quantify vegetation condition or habitat value in temperate Australian ecosystems. These were:

1. Newsome and Catling’s (1979) Habitat Complexity Score (HCS$_N$);
2. Watson et al.’s (2001) Habitat Complexity Score (HCS$_W$);
3. The Site Condition Score (SCS$_{HH}$) component of the Habitat Hectares Index of Parkes et al. (2003);
4. The Vegetation Condition Score (VCS$_{BB}$) component of the Biodiversity Benefits Index of Oliver and Parkes (2003);
5. The Vegetation Condition Score (VCS$_{BM}$) component of the BioMetric Assessment Tool of Gibbons et al. (2004).

I used field data to apply each of these indices to my 48 study sites, and compared the scoring and ranking of sites with my proposed index. I also used these data as a basis for comparing the discrimination power of the various indices.

There were relatively weak, although significant (p<0.05) correlations, between the ranking of sites by my proposed index and by Habitat Complexity Scores (HCS$_N$ $r = 0.38$, and HCS$_W$ $r = 0.39$). These indices clustered sites at the low end of the index scale, and scored a high proportion of sites with the same value. I attributed the relatively poor performance of HCS$_N$ and HCS$_W$ to the limited number of attributes included in these indices, the scoring of attributes on an absolute scale rather than relative to the expected levels for a given vegetation community, and the use of broad classes to score attributes which masked information contained in the field data.

There was an improved correlation with my index, in the ranking of sites by the other three indices (SCS$_{HH}$ $r = 0.52$, VCS$_{BB}$ $r = 0.62$, VCS$_{BM}$ $r = 0.66$). This reflected the inclusion in these indices of attributes describing compositional diversity, coarse woody debris, regeneration, large trees and hollow trees, and the scoring of attributes on a relative scale using benchmarks established for each vegetation community.

Compared to the mean for my index (60%), these indices awarded high scores to sites (SCS$_{HH}$ 77%, VCS$_{BB}$ 74%, VCS$_{BM}$ 68%). These indices also awarded the same score to groups of sites, whereas my index differentiated between these sites. I attributed this
to the wide classes used to score attributes in these indices, which resulted in a disproportionate number of sites falling within the benchmark range for an attribute.

As an objective measure of the performance of all indices I compared their discrimination power in terms of two parameters:

1. The average distance between sites on the index scale (large scale discrimination);
2. The percentage of sites with unique index values (fine scale discrimination).

On this basis my proposed index performed well compared to the other five indices. At a large scale, its discrimination was comparable to the VCS\textsubscript{BM} and SCS\textsubscript{HH}, which had high values for this parameter. This reflected the use of a comprehensive set of attributes in my index and their scoring relative to the range of values occurring in the two different vegetation communities. At a fine scale, my index performed better than all other indices and discriminated between 100\% of sites. This demonstrated the effectiveness of using continuous functions rather than broad classes to score attributes.

### 8.1 Introduction

There are currently two types of indices used to used quantify fauna habitat or vegetation condition in temperate Australian ecosystems; those which score attributes on an absolute scale independent of vegetation community, and those which score attributes relative to benchmarks established for each vegetation community. In this chapter I compare the performance of my proposed index with prominent indices associated with each of these approaches. To do this, I calculated scores for the different indices for each of my 48 study sites using data collected as part of the original fieldwork component of this study. Scores for all indices were then expressed as a percentage and the scoring and ranking of sites compared with my proposed index. I also used these data as a basis for comparing the discrimination power of the various indices.

Ideally this analysis should be based on an independent data set from a new suite of sites, rather than using data from my original study sites, which could favour the performance of my index in a comparison with other indices. This was not a feasible option due to time and resource constraints, and consequently the results of this analysis should be treated cautiously. However,
I partly addressed this concern, by using data from my study sites to develop the benchmarks required to apply three of the indices that were tested. These three indices therefore shared a similar a dependency on my study sites, as my own index.

8.2 Indices which score attributes on an absolute scale

There are two prominent indices that have applied an absolute scale to score structural attributes in temperate Australian ecosystems – Newsome and Catling’s (1979) Habitat Complexity Score, and Watson et al.’s (2001) Habitat Complexity Score. These indices were originally designed as tools for rapidly assessing the habitat requirements of specific faunal groups, and do not include some key attributes which are important to other faunal groups. The indices are quick and easy to use in the field, because they utilise a limited set of attributes, are scored according to set cover classes, and can be applied irrespective of vegetation community.

8.2.1 Habitat Complexity Score (Newsome and Catling, 1979)

Newsome and Catling’s Habitat Complexity Score (HCS\textsubscript{N}) was designed as a tool for rapidly assessing small mammal habitat, and has been correlated with the abundance of ground dwelling mammals (Catling et al., 2000; Catling and Burt, 1995). It incorporates five attributes considered to be important components of small mammal habitat. Each attribute is scored on a scale of 0-3 on the basis of cover classes, and the attribute scores are then summed to give an index value ranging from 0-15 (Table 24). For comparison with my index this value was then converted to a percentage.

When applied to my 48 study sites there was a relatively weak although significant correlation between the ranking of sites by HCS\textsubscript{N} and by my index (Spearman’s $r = 0.38$, $p<0.05$). However, unlike my index, HCS\textsubscript{N} had a skewed distribution amongst study sites, and scored most sites with low index values. For example, HCS\textsubscript{N} scored 44% of sites with the same index value of 40%, yet there were clear differences between these particular sites when assessed in terms of the attributes included in my index (Figure 20).
Table 24: Habitat Complexity Score (Newsome and Catling, 1979). Attributes are scored 0-3 on the basis of cover classes. The same classes are applied irrespective of vegetation community. Scores are summed to give an index value ranging from 0-15.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Score 0</th>
<th>Score 1</th>
<th>Score 2</th>
<th>Score 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Canopy cover (%)</td>
<td>0</td>
<td>&lt;30</td>
<td>30-70</td>
<td>&gt;70</td>
</tr>
<tr>
<td>Shrub cover (%)</td>
<td>0</td>
<td>&lt;30</td>
<td>30-70</td>
<td>&gt;70</td>
</tr>
<tr>
<td>Ground Herbage cover (%)</td>
<td>Sparse</td>
<td>Sparse</td>
<td>Dense</td>
<td>Dense</td>
</tr>
<tr>
<td>Debris cover (logs, rocks) (%)</td>
<td>0</td>
<td>&lt;30</td>
<td>30-70</td>
<td>&gt;70</td>
</tr>
<tr>
<td>Soil moisture</td>
<td>Dry</td>
<td>Moist</td>
<td>Permanent water adjacent</td>
<td>Waterlogged</td>
</tr>
</tbody>
</table>

The relatively poor performance of HCS_N reflected three factors. First, HCS_N scored cover on an absolute scale of 0-100% rather than relative to the expected levels for a given vegetation community. Consequently in the dry communities studied many sites scored low values because these communities have inherently low levels of cover for some attributes. Second, HCS_N uses broad cover classes to score attributes and these classes masked information contained in the original field data. This point is demonstrated in Table 25, which shows the percentage of sites that occurred in each of the four classes suggested by Newsome and Catling (1979), and Watson et al. (2001) for assessing canopy cover as part of a Habitat Complexity Score. In the Newsome and Catling (1979) approach the first and forth classes was redundant and the third class accounted for 73% of sites. Third, a number of the key attributes identified in Chapter 6 (Table 17), such as lifeform and species richness, abundance of overstorey regeneration, overstorey stem size, and the abundance of hollow bearing and dead trees are not included in HCS_N, and this further reduced the ability of this index to differentiate between sites. While the omission of these attributes is not surprising given the original purpose of the index, one must question its characterisation in some studies as a more general surrogate for fauna habitat (e.g. Coops and Catling 1997, 2000). HCS_N is essentially a measure of vegetation biomass and would therefore be more appropriately a termed vegetation biomass score, rather than an index of structural complexity.
Figure 20: Newsome and Catling’s (1979) Habitat Complexity Score (HCS$_N$) had a skewed distribution when applied to 32 dry sclerophyll forest study sites and 16 woodland study sites (a). It scored 44% of sites with the same index value (■) whereas my proposed index successfully differentiated between these particular sites (b).
Table 25. Percentage of 48 study sites (dry sclerophyll and woodland sites combined) contained in each of the four canopy cover classes suggested by Newsome and Catling (1979), and Watson *et al.* (2001), as a component of Habitat Complexity Score. * Indicates a cover class that is redundant.

<table>
<thead>
<tr>
<th>Newsome and Catling (1979) (projective cover %)</th>
<th>Watson <em>et al.</em> (2001) (projective cover %)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cover class</td>
<td>Percentage of sites</td>
</tr>
<tr>
<td>0*</td>
<td>0</td>
</tr>
<tr>
<td>0-30</td>
<td>27%</td>
</tr>
<tr>
<td>30-70</td>
<td>73%</td>
</tr>
<tr>
<td>&gt;70*</td>
<td>0</td>
</tr>
</tbody>
</table>

8.2.2 Habitat Complexity Score (*Watson *et al.*, 2001)

Watson *et al.* (2001) modified HCS\(_N\) in order to characterise bird habitat. Their index (HCS\(_W\)) differs from HCS\(_N\) in that it is based on 6 rather 5 attributes, scores litter and log cover separately, distinguishes between low and tall shrub cover, does not include a soil moisture score, and uses lower thresholds for cover classes in some attributes to reflect the relatively sparse vegetation cover in dry sclerophyll and woodland communities (Table 26). In this modified form, HCS\(_W\) has been significantly correlated with bird species richness (Watson, 1999).

Table 26: Habitat Complexity Score (*Watson *et al.*, 2001). Attributes are scored 0-3 on the basis of cover classes. The same classes are applied irrespective of vegetation community. Scores are summed to give an index value ranging from 0-18.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Score</th>
</tr>
</thead>
<tbody>
<tr>
<td>Canopy cover (%)</td>
<td>0-10</td>
</tr>
<tr>
<td>Tall (2-4m) shrub cover (%)</td>
<td>0-10</td>
</tr>
<tr>
<td>Low (0.5-2m) shrub cover (%)</td>
<td>0-10</td>
</tr>
<tr>
<td>Ground Herbage cover (%)</td>
<td>0-10</td>
</tr>
<tr>
<td>Logs / rocks cover (%)</td>
<td>0-10</td>
</tr>
<tr>
<td>Litter cover (%)</td>
<td>0-10</td>
</tr>
</tbody>
</table>
Despite these changes, \( HCS_W \) performed only slightly better than \( HCS_N \) when applied to my 48 study sites. \( HCS_W \) was only weakly correlated with my proposed index (Spearman’s \( r = 0.39, p < 0.05 \)). It scored most sites with low index values, and failed to distinguish between 54% of sites, all of which scored the same value, although my proposed index clearly differentiated between these particular sites (Figure 21).

The reasons for the poor performance of \( HCS_W \) were the same as for \( HCS_N \), and the lowering of thresholds for cover classes did not improve the discrimination of \( HCS_W \). For example, apart from one exception all sites scored zero for tall shrub cover, low shrub cover and log cover, because they fell within the first cover class. In effect, \( HCS_W \) operated as an index based on just three attributes - canopy cover, ground herbage cover and litter cover.

### 8.3 Indices which score attributes relative to benchmarks

In contrast to Habitat Complexity Scores, benchmarking indices score structural complexity on a relative rather than absolute scale, with stands from different vegetation communities scored relative to different benchmarks. Within a particular vegetation community, attributes for a given stand or patch, are compared to benchmark levels indicative of stands relatively unmodified by post-European landuse. The inherent variation that occurs between stands is addressed by scoring attributes in terms of classes defined by a range either side of the benchmark.
Figure 21: When applied to 32 dry sclerophyll forest study sites and 16 woodland study sites (a), Watson et al.’s (2001) Habitat Complexity Score (HCS<sub>W</sub>) scored 54% of sites with the same index value (■). My proposed index differentiated between these particular sites (b).
As applied by Parkes et al. (2003) and Oliver and Parkes (2003) the benchmarking approach is fundamentally different to the one used in my index. I scored attributes relative to the range of values occurring in all stands, modified and largely unmodified, and did not assume that any particular stand structure was preferable to any other, whereas these authors scored attributes relative to benchmarks indicative of mature, relatively undisturbed stands. Landres et al. (1999) warn against making this type of a priori decision, to benchmark against a particular scale of temporal variation. My approach made no such decision, and acknowledged that regrowth and mid-stage successional stands could contribute to biodiversity because they increase the variety of stand structures present in the landscape. I also acknowledged that some early successional stages may be quite different from older late succession stands but still be “unmodified natural”, and that high levels of structural complexity can occur in stands with intermediate levels of disturbance (e.g. Figure 13, Chapter 7).

Unlike Parkes et al. (2003) and Oliver and Parkes (2003), Gibbons et al. (2004) do not include a preferred successional stage as a basis for establishing their benchmarks. These are constructed on the basis of attribute levels in stands with least post-European modification, which may include regrowth and mid-stage successional stands. The problem with this approach is that stands with quite different structures can contribute to establishing benchmark ranges for attributes, which as a consequence can become so wide that the effectiveness of some attributes in differentiating between stands is reduced.

There are three prominent benchmarking indices that have been applied to temperate Australian ecosystems. These indices incorporate both landscape and stand scale measures of structural complexity, so that to provide a meaningful comparison with my index, I have limited my critique to the stand scale component of each index. These were:

- The Site Condition Score component of the Habitat Hectares Index of Parkes et al. (2003); an index designed by the Victorian Department of Natural Resources and Environment, to quantify vegetation condition as part of a market-based mechanism to improve the quality of native
remnant vegetation;

- The Vegetation Condition Score component of the Biodiversity Benefits Index of Oliver and Parkes (2003); a prototype index proposed by the NSW Department of Infrastructure, Planning and Natural Resources, for assessing the biodiversity benefits or losses resulting from land use change;

- The Vegetation Condition Score component of the BioMetric Assessment Tool of Gibbons et al. (2004); a prototype tool developed by the NSW Department of Environment and Conservation for assessing whether a proposal for broadscale clearing of native vegetation in a Property Vegetation Plan improves or maintains environmental outcomes for biodiversity.

In terms of attributes, there was more agreement between my index and the three benchmarking indices, than there was with Habitat Complexity Scores. As a group, the benchmarking indices contained attributes describing compositional diversity, coarse woody debris, regeneration, large trees and hollow trees (Table 27). There were however a number of important differences in the way my proposed index and the three benchmarking indices assessed these attributes:

- Unlike Parkes et al. (2003), Oliver and Parkes (2003) or Gibbons et al. (2004), who assess the cover of weeds directly, I found it was simpler and more efficient to quantify weeds indirectly through their effect on native plant diversity. I reasoned that when weeds become invasive or threatening they simplify ecosystems by occupying niches that once supported native species, and this will be reflected by a decline in the richness of native lifeforms and perennial species.

- Like Oliver and Parkes (2003) I included the contribution of non-indigenous species to vegetation cover. However, unlike Oliver and Parkes (2003), I did not apply a discount to this contribution, because there was no objective basis for doing this, and for some faunal species exotic cover can be beneficial (Evans, 2000; French and Zubovich, 1997). Gibbons et al. (2004) did not include the contribution of non-indigenous species to vegetation cover in their index because it was
designed to support the NSW Native Vegetation Act (2003), a piece of legislation that is concerned only with indigenous native vegetation.

- Like Gibbons et al., (2004) I limited the direct assessment of regeneration to long-lived overstorey species, whereas Parkes et al. (2003) and Oliver and Parkes (2003) also assessed regeneration in understorey woody species. This can be difficult in practice, because of the small size of understorey regeneration, its sparse cover and its often similar growth form to adult plants. I decided that the most efficient way to monitor understorey regeneration was indirectly through its effects on vegetation cover, lifeform richness and perennial species richness.

- Rather than estimate canopy cover directly, as do Parkes et al. (2003) Oliver and Parkes (2003) and Gibbons et al. (2004), I used the surrogate stand basal area of live trees. This was because the stand basal area of live trees was well correlated with canopy cover, could be measured more precisely than canopy cover, and was correlated with a greater number of additional attributes than was the case for canopy cover.

- Unlike Parkes et al. (2003) and Oliver and Parkes (2003) I quantified litter in terms of biomass (dry weight) rather than cover, which appeared to be a poor measure in dry sclerophyll and woodland communities (Figure 8, Chapter 6). Gibbons et al. (2004) appear to concur with this assessment because they exclude litter cover as an attribute in their index.

- I included the additional attributes of quadratic mean diameter and the number of dead trees, because my data set confirmed these were important attributes, which differentiated between sites.

There were also differences in the way my index scored attributes, and combined these scores in an index value. The three benchmarking indices scored attributes on the basis of classes, which lost information contained in field data, and masked differences between sites. My index scored attributes using continuous functions, which maintained a high correlation between attribute scores and the relevant field data.
Table 27. Comparison between the 13 attributes included in my proposed index and attributes included in the stand scale component of three indices currently used to assess vegetation condition in temperate Australian ecosystems, and which score attributes relative to benchmarks (A Stand component of Habitat Hectares Index; B Stand component of Biodiversity Benefits Index; C Stand component of BioMetric assessment tool). For each index, the weightings applied to attributes are shown as a percentage.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Index proposed in this thesis</th>
<th>Site Condition Score&lt;sup&gt;A&lt;/sup&gt; (Parkes et al., 2003)</th>
<th>Vegetation Condition Score&lt;sup&gt;B&lt;/sup&gt; (Oliver &amp; Parkes, 2003)</th>
<th>Vegetation Condition Score&lt;sup&gt;C&lt;/sup&gt; (Gibbons et al., 2004)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Perennial species richness (native)</td>
<td>7.7</td>
<td>7.7</td>
<td>Richness of native species within lifeforms</td>
<td>Native plant species richness</td>
</tr>
<tr>
<td>Lifeform richness (native)</td>
<td>7.7</td>
<td></td>
<td>25</td>
<td>20</td>
</tr>
<tr>
<td>Vegetation cover &gt;0.5m</td>
<td>7.7</td>
<td>7.7</td>
<td>Richness of native species within lifeforms assessed concurrently with Cover of native understorey lifeforms</td>
<td>33</td>
</tr>
<tr>
<td>Vegetation cover &lt;0.5m</td>
<td>7.7</td>
<td></td>
<td>20</td>
<td>Native foliage cover (shrubs)</td>
</tr>
<tr>
<td>Stand basal area of live trees</td>
<td>7.7</td>
<td>Tree (canopy) cover</td>
<td>6.7</td>
<td>Native foliage cover (other)</td>
</tr>
<tr>
<td>Number of regenerating overstorey stems</td>
<td>7.7</td>
<td>Presence of adequate regeneration in woody perennial native species</td>
<td>13</td>
<td>Native mid-storey foliage cover</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>10</td>
<td>Native overstorey foliage cover</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Lack of weed cover</td>
<td>20</td>
<td>Lack of exotic plant foliage cover</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Litter dry weight</td>
<td>6.7</td>
<td>5</td>
</tr>
<tr>
<td>Total log length</td>
<td>7.7</td>
<td>Total log length assessed concurrently with Large logs</td>
<td>6.7</td>
<td>Total length of fallen logs</td>
</tr>
<tr>
<td>Total large log length</td>
<td>7.7</td>
<td></td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>Quadratic mean diameter</td>
<td>7.7</td>
<td>Number of large trees</td>
<td>13</td>
<td>Number of trees with hollows</td>
</tr>
<tr>
<td>Number of live stems &gt; 40cm dbh</td>
<td>7.7</td>
<td></td>
<td>15</td>
<td>30</td>
</tr>
<tr>
<td>Number of hollow bearing trees</td>
<td>7.7</td>
<td>Number of large trees</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>Number of dead trees</td>
<td>7.7</td>
<td></td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td>100</td>
<td>Total</td>
<td>100</td>
<td>Total</td>
</tr>
</tbody>
</table>

Total 100
Within each of the three benchmarking indices, attributes were differentially weighted (Table 27). The size of these weightings appeared a matter of subjective judgement, which reflected the relative value the authors placed on each attribute, rather than any quantitative assessment of the effect weighting had on the performance of the index. For example Gibbons et al., (2004) gave hollow bearing trees a weight of 30% whereas Oliver and Parkes (2003) applied a weighting of 5% and Parkes et al. (2003) excluded this attribute altogether. In my index, attributes carried equal weight, because sensitivity analysis indicated that the performance of my index was independent of the weighting of attributes.

### 8.3.1 Site Condition Score - Habitat Hectares Index (Parkes et al., 2003)

To apply the Site Condition Score (SCS$_{HH}$) component of the Habitat Hectares Index to my study sites, the necessary benchmarks were established using data from 16 high quality dry sclerophyll forest sites, and 8 high quality woodland sites. These sites were relatively unmodified compared to other sites. The mean levels for the benchmarks established this way differed considerably from those proposed for comparable communities in Victoria (Table 28). This discrepancy may reflect real differences between the communities, or the fact that many of the Victorian Benchmarks were established on the basis of expert opinion rather than actual field data (Phil Gibbons pers. comm., 2004).

Table 28: Comparison between benchmarks established using data from 16 high quality dry sclerophyll forest sites, and 8 high quality woodland sites, and benchmarks proposed for comparable ecological vegetation communities (EVC) in Victoria on the basis of expert opinion.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Woodland sites (Data)</th>
<th>Grass woodland EVC (Expert opinion)</th>
<th>Dry sclerophyll forest sites (Data)</th>
<th>Grassy dry forest EVC (Expert opinion)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Canopy cover (%)</td>
<td>31</td>
<td>20</td>
<td>44</td>
<td>30</td>
</tr>
<tr>
<td>Large trees (stems ha$^{-1}$)</td>
<td>18</td>
<td>6</td>
<td>9</td>
<td>13</td>
</tr>
<tr>
<td>Understorey lifeforms (per 400m$^2$)</td>
<td>5</td>
<td>5</td>
<td>5</td>
<td>4</td>
</tr>
<tr>
<td>Litter cover (%)</td>
<td>66</td>
<td>30</td>
<td>79</td>
<td>70</td>
</tr>
<tr>
<td>Log length (mha$^{-1}$)</td>
<td>308</td>
<td>150</td>
<td>643</td>
<td>100</td>
</tr>
</tbody>
</table>
Attribute scores in the SCS\textsubscript{HH} are determined by how much each attribute differs from its benchmark value. Broad class intervals are used for this assessment in an attempt to reflect the inherent range of variation in unmodified stands (Parkes et al., 2003). These class intervals were established on the basis of expert opinion rather than measured variation in real stands. When applied to my study sites the width of these intervals masked differences between sites, and awarded high scores to a disproportionate number of sites. For example in the SCS\textsubscript{HH}, the maximum score for canopy cover is 5, and is awarded to sites with ± 50\% of the benchmark canopy cover value. When applied to my study sites this resulted in 77\% of sites scoring a 5 for canopy cover, even though this attribute was normally distributed and ranged from 8-62\% cover (Figure 22). Different class interval widths are used for different attributes in the SCS\textsubscript{HH}, so that the scoring system can be complex to apply in practice. This is highlighted in Table 29, which details how the various benchmarks and associated class intervals were used to score my study sites.

When applied to my 48 study sites, the performance of the SCS\textsubscript{HH} was an improvement on Habitat Complexity Scores. The SCS\textsubscript{HH} better differentiated between sites as evidenced by an approximately normal distribution of scores amongst sites (Figure 23). There was also an improved correlation with my index, in the ranking of sites by SCS\textsubscript{HH} (\(r = 0.52\), \(p<0.05\)), compared to rankings using Habitat Complexity Scores (HCS\textsubscript{N} \(r = 0.38\), HCS\textsubscript{W} \(r = 0.39\)). The SCS\textsubscript{HH} correlation increased to 0.62 if one particular site was excluded. This site scored high on my index because it contained a large number of hollow-bearing trees, and high levels of ground vegetation cover due to largely exotic species. This site scored low on the SCS\textsubscript{HH}, because it did not quantify these attributes. However, overall the SCS\textsubscript{HH} awarded high scores to sites with a mean score of 77\%, compared to the mean for my index of 60\%. These high scores reflected the wide classes used to score attributes in the SCS\textsubscript{HH}, which resulted in a disproportionate number of sites falling within the benchmark range for an attribute. This effect, combined with the absence of some key attributes (hollow-bearing trees, dbh\textsubscript{Q}, dead trees), meant that SCS\textsubscript{HH} tended to award similar high total scores to groups of sites; whereas my index indicated there were distinct differences between these sites (Figure 23).
Figure 22: The use of broad classes to score attributes in the Site Condition Score of Parkes et al. (2003) masked differences between sites and awarded most sites with high scores. Demonstrated here for the canopy cover component of the Site Condition Score: (a) the distribution of canopy cover amongst 16 woodland and 32 dry sclerophyll forest study sites (b) the distribution of canopy cover scores for the same sites.
Table 29: Benchmarks and class intervals, used to score 16 woodland and 32 dry sclerophyll forest study sites according to the Site Condition Score component of the Habitat Hectares Index of Parkes et al. (2003). Total possible score was 75, benchmark values are in bold, ( ) indicates maximum score for an attribute, scores for different classes in bold italics. A Canopy health assessed in terms of proportion of expected canopy cover missing due to tree death decline or mistletoe infection. B A lifeform was modified if it had <50% of benchmark richness, or in the case of Tussock Grass <50% of benchmark cover. C Only overstorey recruitment was considered.

<table>
<thead>
<tr>
<th>Woodland sites</th>
<th>Dry sclerophyll forest sites</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Canopy cover (5): 31%</strong></td>
<td><strong>Canopy cover (5): 44%</strong></td>
</tr>
<tr>
<td>Canopy health</td>
<td>Canopy health</td>
</tr>
<tr>
<td>&gt;70%</td>
<td>&gt;70%</td>
</tr>
<tr>
<td>30-70%</td>
<td>30-70%</td>
</tr>
<tr>
<td>&lt;30% or &gt;59%</td>
<td>&lt;30% or &gt;84%</td>
</tr>
<tr>
<td>&lt;3-15.5% or 46.5-59%</td>
<td>4-22% or 66-84%</td>
</tr>
<tr>
<td>15.5-46.5%</td>
<td>22-66%</td>
</tr>
<tr>
<td></td>
<td>0 0 0</td>
</tr>
<tr>
<td></td>
<td>0 0 0</td>
</tr>
<tr>
<td></td>
<td>3 2 1</td>
</tr>
<tr>
<td></td>
<td>3 2 1</td>
</tr>
<tr>
<td></td>
<td>5 4 3</td>
</tr>
<tr>
<td></td>
<td>5 4 3</td>
</tr>
<tr>
<td><strong>Large trees (10): 18 ha(^1)</strong></td>
<td><strong>Large trees (10): 9 ha(^1)</strong></td>
</tr>
<tr>
<td>Canopy health</td>
<td>Canopy health</td>
</tr>
<tr>
<td>&gt;70%</td>
<td>&gt;70%</td>
</tr>
<tr>
<td>30-70%</td>
<td>30-70%</td>
</tr>
<tr>
<td>&lt;30% or &gt;59%</td>
<td>&lt;4-8 ha(^1)</td>
</tr>
<tr>
<td>&lt;3-15.5% or 46.5-59%</td>
<td>0-1.8 ha(^1)</td>
</tr>
<tr>
<td>15.5-46.5%</td>
<td>1.8-3.6 ha(^1)</td>
</tr>
<tr>
<td></td>
<td>3.6-6.3 ha(^1)</td>
</tr>
<tr>
<td></td>
<td>6.3-9 ha(^1)</td>
</tr>
<tr>
<td></td>
<td>&gt;9 ha(^1)</td>
</tr>
<tr>
<td></td>
<td>0 0 0</td>
</tr>
<tr>
<td></td>
<td>3 2 1</td>
</tr>
<tr>
<td></td>
<td>4 3 2</td>
</tr>
<tr>
<td></td>
<td>6 5 4</td>
</tr>
<tr>
<td></td>
<td>8 7 6</td>
</tr>
<tr>
<td></td>
<td>10 9 8</td>
</tr>
<tr>
<td><strong>Understorey lifeforms (25): 5 per 400m(^2)</strong></td>
<td><strong>Understorey lifeforms (25): 7 per 400m(^2)</strong></td>
</tr>
<tr>
<td>All lifeforms absent</td>
<td>All lifeforms absent</td>
</tr>
<tr>
<td>&lt;3 lifeforms present</td>
<td>&lt;4 lifeforms present</td>
</tr>
<tr>
<td>3-4 lifeforms present</td>
<td>4-6 lifeforms present</td>
</tr>
<tr>
<td>≥2 modified(^a)</td>
<td>≥3 modified(^a)</td>
</tr>
<tr>
<td>&lt;2 modified(^a)</td>
<td>&lt;3 modified(^a)</td>
</tr>
<tr>
<td>&gt;2 modified(^a)</td>
<td>≥3 modified(^a)</td>
</tr>
<tr>
<td>none modified(^a)</td>
<td>none modified(^a)</td>
</tr>
<tr>
<td></td>
<td>5 10</td>
</tr>
<tr>
<td></td>
<td>10 15</td>
</tr>
<tr>
<td></td>
<td>20 25</td>
</tr>
<tr>
<td><strong>Weed cover (15)</strong></td>
<td><strong>Weed cover (15)</strong></td>
</tr>
<tr>
<td>High threat weed proportion</td>
<td>High threat weed proportion</td>
</tr>
<tr>
<td>≥50% cover of weeds</td>
<td>≥50% cover of weeds</td>
</tr>
<tr>
<td>25-50% cover of weeds</td>
<td>25-50% cover of weeds</td>
</tr>
<tr>
<td>5-25% cover of weeds</td>
<td>5-25% cover of weeds</td>
</tr>
<tr>
<td>&lt;5% cover of weeds</td>
<td>&lt;5% cover of weeds</td>
</tr>
<tr>
<td>0</td>
<td>0 4</td>
</tr>
<tr>
<td>≤50%</td>
<td>2 2</td>
</tr>
<tr>
<td>&gt;50%</td>
<td>0 4</td>
</tr>
<tr>
<td>11</td>
<td>7 1</td>
</tr>
<tr>
<td>15</td>
<td>9 7</td>
</tr>
<tr>
<td>13</td>
<td>11 11</td>
</tr>
<tr>
<td><strong>Recruitment(^c) (10): 310 stems ha(^{-1})</strong></td>
<td><strong>Recruitment(^c) (10): 210 stems ha(^{-1})</strong></td>
</tr>
<tr>
<td>≥2 species</td>
<td>≥2 species</td>
</tr>
<tr>
<td>1 species</td>
<td>1 species</td>
</tr>
<tr>
<td>None</td>
<td>None</td>
</tr>
<tr>
<td>&lt;90 stems ha(^{-1})</td>
<td>&lt;60 stems ha(^{-1})</td>
</tr>
<tr>
<td>90-215 stems ha(^{-1})</td>
<td>60-150 stems ha(^{-1})</td>
</tr>
<tr>
<td>&gt;215 stems ha(^{-1})</td>
<td>&gt;150 stems ha(^{-1})</td>
</tr>
<tr>
<td>0</td>
<td>0 4</td>
</tr>
<tr>
<td>3</td>
<td>3 1</td>
</tr>
<tr>
<td>6</td>
<td>6 3</td>
</tr>
<tr>
<td>10</td>
<td>10 15</td>
</tr>
<tr>
<td><strong>Litter cover (5): 66%</strong></td>
<td><strong>Litter cover (5): 79%</strong></td>
</tr>
<tr>
<td>&lt;7% cover</td>
<td>&lt;8%</td>
</tr>
<tr>
<td>7-33% cover</td>
<td>8-40%</td>
</tr>
<tr>
<td>33-100% cover</td>
<td>40-100%</td>
</tr>
<tr>
<td>0</td>
<td>0 4</td>
</tr>
<tr>
<td>3</td>
<td>3 3</td>
</tr>
<tr>
<td>5</td>
<td>5 5</td>
</tr>
<tr>
<td><strong>Log length (5): 308 mha(^{-1})</strong></td>
<td><strong>Log length (5): 643 mha(^{-1})</strong></td>
</tr>
<tr>
<td>≥25% large logs</td>
<td>≥25% large logs</td>
</tr>
<tr>
<td>&lt;25% large logs</td>
<td>&lt;25% large logs</td>
</tr>
<tr>
<td>&lt; 31 mha(^{-1})</td>
<td>&lt; 64 mha(^{-1})</td>
</tr>
<tr>
<td>31-154 mha(^{-1})</td>
<td>64-322 mha(^{-1})</td>
</tr>
<tr>
<td>154-462 mha(^{-1})</td>
<td>322-965 mha(^{-1})</td>
</tr>
<tr>
<td>&gt;462 mha(^{-1})</td>
<td>&gt;965 mha(^{-1})</td>
</tr>
<tr>
<td>0</td>
<td>0 0</td>
</tr>
<tr>
<td>3</td>
<td>3 2</td>
</tr>
<tr>
<td>5</td>
<td>5 4</td>
</tr>
<tr>
<td>3</td>
<td>3 2</td>
</tr>
</tbody>
</table>
Chapter 8: Comparing the performance of the index to other indices

Figure 23: The Site Condition Score ($SCS_{HH}$) component of the Habitat Hectares Index of Parkes et al. (2003) had an approximately normal distribution when applied to 32 dry sclerophyll forest study sites and 16 woodland study sites (a). It awarded similar high scores to groups of sites (■), whereas my proposed index differentiated between these particular sites (b).
8.3.2 Vegetation Condition Score – Biodiversity Benefits Index (Oliver and Parkes, 2003)

Oliver and Parkes (2003) developed their Vegetation Condition Score (VCS$_{BB}$) directly from the SCS$_{HH}$, and incorporated three important changes. First, they assessed the richness and cover of plant groups separately rather than together. Richness is assessed in terms of native species present in five different plant groups (trees, shrubs, forbs, perennial grasses, other) and for cover a sixth group (cryptogams) is also assessed. The contribution to cover of exotic and non-indigenous native species vegetation is included, although at a discounted rate. Oliver and Parkes (2003) give each plant group equal weight in the final richness or cover score. However, no rationale is given for this approach, which for some groups seems counter-intuitive. For example it is unclear why cryptogam cover would contribute a comparable amount to biodiversity as tree, shrub or perennial grass cover.

The second change was to use five different benchmarks instead of one to score the richness and cover of plant groups. The intention was to capture the range of condition (very low, low, moderate, high and very high), occurring in unmodified stands. However, Oliver and Parkes (2003) do not indicate the basis for establishing these different categories of benchmark, or make it clear exactly what these categories mean. For example, it appears contradictory to categorise an unmodified stand as having very low condition, because this implies it is at less than its potential condition, and so must have been modified.

The third change was the inclusion of the attribute, density of hollow-bearing trees, in addition to the density of large trees. This decision is supported by data from my study sites (Chapter 6), and by Gibbons and Lindenmayer (2002), who indicate that the majority of hollow-bearing trees in a stand usually occur in the middle diameter classes, rather than in the largest trees.

To apply the Vegetation Condition Score (VCS$_{BB}$) to my study sites, I followed the same procedure I used with Parkes et al.’s SCS$_{HH}$, and established benchmarks using data from 16 dry sclerophyll forest sites, and 8 woodland
sites, which were relatively unmodified compared to other sites. In the absence of any clear guide as to how to establish the different categories of benchmarks for scoring the richness and cover of plant groups, I used the 0-20, 20-40, 40-60, 60-80, 80-100 percentiles for the relevant attributes in my data set. I did not follow this approach for tree cover, because Oliver and Parkes (2003) consider the average, rather than the highest level of tree cover to be optimum. For tree cover I therefore equated the “very high” condition to the 40-60 percentile range, “high” the 60-80, “moderate” the 20-40, “low” the 80-100, and “very low” the “0-20. The resulting benchmarks and associated class intervals are shown in Table 30, and were used to calculate a VCS_{BB} for each of my 48 study sites.

The performance of the VCS_{BB} was an improvement on SCS_{HH} when applied to my study sites. VCS_{BB} was approximately normally distributed (Figure 24), and in terms of the ranking of sites had a higher correlation with my index (r = 0.62), than did SCS_{HH} (r = 0.52). The site, which was an outlier when assessed by SCS_{HH} was no longer one, because VCS_{BB} included additional attributes describing hollow bearing trees and exotic vegetation cover which were critical for characterising this particular site.

However, because the VCS_{BB} was developed directly from SCS_{HH} there were a number of similarities in the performance of the two indices. In particular, the ranking of sites by VCS_{BB} was highly correlated with the ranking of sites by SCS_{HH} (r = 0.81); the VCS_{BB} used classes to score attributes and so like the SCS_{HH} tended to score sites higher than my index did, with the mean score for VCS_{BB} 74% compared to the mean score for my index of 60%; and like the SCS_{HH} the VCS_{BB} awarded groups of sites similar high scores, while my proposed index differentiated between these particular sites (Figure 24).
Chapter 8: Comparing the performance of the index to other indices

Table 30: Benchmarks and class intervals, used to score 16 woodland and 32 dry sclerophyll forest study sites according to the Vegetation Condition Score component of the Biodiversity Benefits Index of Oliver and Parkes (2003). Total possible score was 100, benchmark values are in bold, ( ) indicates maximum score for an attribute, scores for different classes in bold italics.

- Richness and cover scores were the average of scores for the various plant groups.
- Only overstorey recruitment was considered.
- Canopy health assessed in terms of proportion of expected canopy cover missing due to tree death decline or mistletoe infection.
- Litter depth assumed to be > 1 cm.

<table>
<thead>
<tr>
<th>Woodland sites</th>
<th>Dry sclerophyll forest sites</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Richness</strong>&lt;sup&gt;A&lt;/sup&gt; (25): species per 400 m&lt;sup&gt;2&lt;/sup&gt; plot</td>
<td><strong>Richness</strong>&lt;sup&gt;A&lt;/sup&gt; (25): species per 400 m&lt;sup&gt;2&lt;/sup&gt; plot</td>
</tr>
<tr>
<td>5  10  15  20  25</td>
<td>5  10  15  20  25</td>
</tr>
<tr>
<td>Trees &lt;2 3 4 4 (&gt;4)</td>
<td>Trees 1 2 3 4 (&gt;4)</td>
</tr>
<tr>
<td>Shrubs 0 1 2 3-5 (&gt;5)</td>
<td>Shrubs &lt;5 5-7 8-9 10-11 (&gt;11)</td>
</tr>
<tr>
<td>Perenn. grass &lt;3 3-4 5 6 (&gt;6)</td>
<td>Perenn. grass 1 2 2 3 (&gt;3)</td>
</tr>
<tr>
<td>Other 1 2 3 4 (&gt;4)</td>
<td>Other 1 2 3 4 (&gt;4)</td>
</tr>
<tr>
<td><strong>Cover</strong>&lt;sup&gt;A&lt;/sup&gt; (20): % cover per 400 m&lt;sup&gt;2&lt;/sup&gt; plot</td>
<td><strong>Cover</strong>&lt;sup&gt;A&lt;/sup&gt; (20): % cover per 400 m&lt;sup&gt;2&lt;/sup&gt; plot</td>
</tr>
<tr>
<td>5  10  15  20  25</td>
<td>5  10  15  20  25</td>
</tr>
<tr>
<td>Shrubs 0 0 &lt;1 1-3 (&lt;3)</td>
<td>Shrubs &lt;2 2-5 5-6 6-10 (&gt;10)</td>
</tr>
<tr>
<td>Perenn. grass &lt;30 30-40 40-55 55-65 (&gt;65)</td>
<td>Perenn. grass &lt;12 12-25 25-35 35-45 (45)</td>
</tr>
<tr>
<td>Other &lt;1 1-2 2-3 3-4 (4)</td>
<td>Other 0 &lt;1 1-1.5 1.5-3 (&gt;3)</td>
</tr>
<tr>
<td>Cryptogam &lt;1 1-5 5-8 8-14 (&gt;14)</td>
<td>Cryptogam &lt;1 1-3 3-4 4-8 (8)</td>
</tr>
<tr>
<td><strong>Weed cover</strong> (15): % cover per 400 m&lt;sup&gt;2&lt;/sup&gt; plot</td>
<td><strong>Weed cover</strong> (15): % cover per 400 m&lt;sup&gt;2&lt;/sup&gt; plot</td>
</tr>
<tr>
<td>High threat weed proportion &gt;50%</td>
<td>High threat weed proportion ≤50%</td>
</tr>
<tr>
<td>≥50% cover of weeds 4 2 0</td>
<td>≥50% cover of weeds 4 2 0</td>
</tr>
<tr>
<td>5-50% cover of weeds 7 6 4</td>
<td>25-50% cover of weeds 7 6 4</td>
</tr>
<tr>
<td>5-25% cover of weeds 11 9 7</td>
<td>5-25% cover of weeds 11 9 7</td>
</tr>
<tr>
<td>&lt;5% cover of weeds 15 13 11</td>
<td>&lt;5% cover of weeds 15 13 11</td>
</tr>
<tr>
<td><strong>Recruitment</strong>&lt;sup&gt;B&lt;/sup&gt; (10): 310 stems ha&lt;sup&gt;-1&lt;/sup&gt;</td>
<td><strong>Recruitment</strong>&lt;sup&gt;B&lt;/sup&gt; (10): 210 stems ha&lt;sup&gt;-1&lt;/sup&gt;</td>
</tr>
<tr>
<td>≥2 species</td>
<td>≥2 species</td>
</tr>
<tr>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>1-2 species</td>
<td>1-2 species</td>
</tr>
<tr>
<td>3 1</td>
<td>3 1</td>
</tr>
<tr>
<td>None</td>
<td>None</td>
</tr>
<tr>
<td>0 0</td>
<td>0 0</td>
</tr>
<tr>
<td>1-2 species</td>
<td>1-2 species</td>
</tr>
<tr>
<td>10 5</td>
<td>10 5</td>
</tr>
<tr>
<td><strong>Large trees</strong> (15): 18 stems ha&lt;sup&gt;-1&lt;/sup&gt;</td>
<td><strong>Large trees</strong> (15): 9 stems ha&lt;sup&gt;-1&lt;/sup&gt;</td>
</tr>
<tr>
<td>&gt;70%</td>
<td>&gt;70%</td>
</tr>
<tr>
<td>Canopy health 0 0 0</td>
<td>Canopy health 0 0 0</td>
</tr>
<tr>
<td>30-70%</td>
<td>30-70%</td>
</tr>
<tr>
<td>6 4 2</td>
<td>6 4 2</td>
</tr>
<tr>
<td>3-4</td>
<td>3-4</td>
</tr>
<tr>
<td>3-5</td>
<td>3-5</td>
</tr>
<tr>
<td>3-7</td>
<td>3-7</td>
</tr>
<tr>
<td>7-12</td>
<td>7-12</td>
</tr>
<tr>
<td>12-16</td>
<td>12-16</td>
</tr>
<tr>
<td>&gt;16</td>
<td>&gt;16</td>
</tr>
<tr>
<td>None</td>
<td>None</td>
</tr>
<tr>
<td>0 0 0</td>
<td>0 0 0</td>
</tr>
<tr>
<td>0.3-6 ha&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>0.3-6 ha&lt;sup&gt;-1&lt;/sup&gt;</td>
</tr>
<tr>
<td>6 4 2</td>
<td>6 4 2</td>
</tr>
<tr>
<td>3.6-7.2 ha&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>3.6-7.2 ha&lt;sup&gt;-1&lt;/sup&gt;</td>
</tr>
<tr>
<td>8 7 5</td>
<td>8 7 5</td>
</tr>
<tr>
<td>7.2-12.6 ha&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>7.2-12.6 ha&lt;sup&gt;-1&lt;/sup&gt;</td>
</tr>
<tr>
<td>12 10 8</td>
<td>12 10 8</td>
</tr>
<tr>
<td>&gt;12.6 ha&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>&gt;12.6 ha&lt;sup&gt;-1&lt;/sup&gt;</td>
</tr>
<tr>
<td>15 13 11</td>
<td>15 13 11</td>
</tr>
<tr>
<td><strong>Hollow bearing trees</strong> (5): 14 trees ha&lt;sup&gt;-1&lt;/sup&gt;</td>
<td><strong>Hollow bearing trees</strong> (5): 60 trees ha&lt;sup&gt;-1&lt;/sup&gt;</td>
</tr>
<tr>
<td>0 2 3 4 5</td>
<td>0 2 3 4 5</td>
</tr>
<tr>
<td>0 0-3 3-6 6-10 &gt;10</td>
<td>0 0-12 12-24 24-42 &gt;42</td>
</tr>
<tr>
<td><strong>Litter</strong> (5): 66%</td>
<td><strong>Litter</strong> (5): 79%</td>
</tr>
<tr>
<td>&lt;7% cover</td>
<td>&lt;7% cover</td>
</tr>
<tr>
<td>0</td>
<td>&lt;8%</td>
</tr>
<tr>
<td>7-33% cover</td>
<td>8-40%</td>
</tr>
<tr>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>33-100% cover</td>
<td>40-100%</td>
</tr>
<tr>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td><strong>Log length</strong> (5): 308 mha&lt;sup&gt;-1&lt;/sup&gt;</td>
<td><strong>Log length</strong> (5): 643 mha&lt;sup&gt;-1&lt;/sup&gt;</td>
</tr>
<tr>
<td>% in advanced state of decay</td>
<td>% in advanced state of decay</td>
</tr>
<tr>
<td>&lt;25%</td>
<td>&lt;25%</td>
</tr>
<tr>
<td>0 1</td>
<td>0 1</td>
</tr>
<tr>
<td>&gt;25%</td>
<td>&gt;25%</td>
</tr>
<tr>
<td>2 3</td>
<td>2 3</td>
</tr>
<tr>
<td>31-154 mha&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>31-154 mha&lt;sup&gt;-1&lt;/sup&gt;</td>
</tr>
<tr>
<td>4 5</td>
<td>4 5</td>
</tr>
</tbody>
</table>
Figure 24: The Vegetation Condition Score (VCS$_{BB}$) component of the Biodiversity Benefits Index of Oliver and Parkes (2003) had an approximately normal distribution when applied to 32 dry sclerophyll forest study sites and 16 woodland study sites (a). It awarded similar high scores to groups of sites (■), whereas my proposed index differentiated between these particular sites (b).
8.3.3 Vegetation Condition Score – BioMetric Assessment Tool (Gibbons et al., 2004)

Although Gibbons et al. (2004) utilise a number of the key principles underpinning the benchmarking approach of Parkes et al. (2003), their Vegetation Condition Score (VCS<sub>BM</sub>) was not developed directly from either the SCS<sub>HH</sub> of Parkes et al. (2003) and the VCS<sub>BB</sub> of Oliver and Parkes (2003). It differs from these indices in terms of the attributes that are used in the index, the weightings applied to each attribute, and the method for comparing attribute levels with benchmarks.

In terms of attributes, the VCS<sub>BM</sub> does not include litter cover or the number of large trees, the assessment of regeneration is limited to overstorey species rather than all perennial woody species, the contribution of exotic and non-indigenous species to vegetation cover is not considered because the index supports a piece of legislation that is concerned only with indigenous native vegetation, and log size or decay class are not assessed.

In terms of attribute weightings, the VCS<sub>BM</sub> allocates six times the weighting (30%) to the number of hollow-bearing trees compared to the VCS<sub>BB</sub> (5%), one quarter the weighting (5%) to lack of weed cover compared to VCS<sub>BB</sub> (20%), and one third the weighting to lack of weed cover compared to SCS<sub>HH</sub> (15%).

In terms of scoring, the VCS<sub>BM</sub> uses upper and lower benchmark values as a basis for constructing the class intervals used to score the cover of different plant groups. For a given attribute, these benchmarks are defined by the upper (B<sub>max</sub>) and lower (B<sub>min</sub>) boundaries of the 95% confidence interval of the mean in relatively unmodified sites. Four classes (0,1,2,3) are defined using these benchmarks as follows:

0 points \( \leq 0.1*B_{min} \) or \( >2*B_{max} \)
1 point \( 0.1*B_{min} < \) and \( <0.5*B_{min} \) or \( 1.5*B_{max} < \) and \( \leq 2*B_{max} \)
2 points \( 0.5*B_{min} \leq \) and \( <B_{min} \) or \( B_{max} < \) and \( \leq 1.5*B_{max} \)
3 points \( B_{min} \leq \) and \( \leq B_{max} \)

This approach contrasts with the SCS<sub>HH</sub> and VCS<sub>BB</sub>, which do not provide a
clear basis for relating class boundaries to observed variation in reference sites; although the factors (0.1, 0.5, 1.5 and 2) used to create the class boundaries in the VCS\textsubscript{BM} remains a matter of subjective choice.

To apply Gibbons \textit{et al.}’s VCS\textsubscript{BM} to my study sites, I established the necessary benchmarks using data from 16 dry sclerophyll forest sites, and 8 woodland sites, which were relatively unmodified compared to other sites. The resulting benchmarks and associated class intervals are shown in Table 31, and were used to calculate a VCS\textsubscript{BM} for each of my 48 study sites.

The VCS\textsubscript{BM} had a skewed distribution across my study sites, and the average score for sites (68\%) was significantly (p<0.05) higher than the average score awarded by my index (60\%). As was the case with the SCS\textsubscript{HH} and VCS\textsubscript{BB}, these high scores reflected the use of broad classes to score attributes. It appeared that the use of the 95\% confidence interval to define classes for some attributes did not prevent this approach from losing information contained in the original field data (Figure 26). Consequently the VCS\textsubscript{BM} awarded groups of sites similar high scores, while my proposed index differentiated between these particular sites (Figure 27). However in terms of the ranking of sites the VCS\textsubscript{BM} had a higher correlation with my index (r = 0.66) than did either the VCS\textsubscript{BB} (r = 0.62) or the SCS\textsubscript{HH} (r = 0.52).
Table 31: Benchmarks and class intervals, used to score 16 woodland and 32 dry sclerophyll forest study sites, according to the Vegetation Condition Score component of the BioMetric Assessment Tool of Gibbons et al. (2004). Total possible score was 100, benchmark values are in bold, ( ) indicates maximum score for an attribute, scores for different classes in bold italics. \(^A\)Assessed in terms of native perennial species. \(^B\)Projective foliage cover in 400 m\(^2\) plots. \(^D\)Sedges, rushes, ferns, vines and xanthorrhoea. \(^E\)Proportion regenerating.

<table>
<thead>
<tr>
<th>Woodland sites</th>
<th>Dry sclerophyll forest sites</th>
</tr>
</thead>
<tbody>
<tr>
<td>Richness(^A) (20): 14 species per 400 m(^2) plot</td>
<td>Richness(^A) (20): 21 species per 400 m(^2) plot</td>
</tr>
<tr>
<td>None</td>
<td>None</td>
</tr>
<tr>
<td>0(&lt;) and (&lt;)7 species</td>
<td>0(&lt;) and (&lt;)10 species</td>
</tr>
<tr>
<td>7(\leq) and (&lt;)14 species</td>
<td>10(\leq) and (&lt;)21 species</td>
</tr>
<tr>
<td>(\geq)14 species</td>
<td>(\geq)21 species</td>
</tr>
<tr>
<td>Overstorey cover(^B) (5): 31%</td>
<td>Overstorey cover(^B) (5): 44%</td>
</tr>
<tr>
<td>(\leq)2% or (&gt;)82%</td>
<td>(\leq)4% or (&gt;)94%</td>
</tr>
<tr>
<td>2(&lt;) and (&lt;)10% or 62(\leq) and (&lt;)82%</td>
<td>4(&lt;) and (&lt;)21% or 71(&lt;) and (&lt;)94%</td>
</tr>
<tr>
<td>10(\leq) and (&lt;)21% or 41(&lt;) and (&lt;)62%</td>
<td>21(&lt;) and (&lt;)41% or 47(&lt;) and (&lt;)71%</td>
</tr>
<tr>
<td>21(\leq) and (\geq)41%</td>
<td>41(\leq) and (\geq)47%</td>
</tr>
<tr>
<td>Mid-storey cover (5):</td>
<td>Mid-storey cover (5):</td>
</tr>
<tr>
<td>Not applicable to this community - all sites score 5</td>
<td>Not applicable to this community - all sites score 5</td>
</tr>
<tr>
<td>Native grass cover(^B) (5): 43%</td>
<td>Native grass cover(^B) (5): 28%</td>
</tr>
<tr>
<td>(\leq)3%</td>
<td>(\leq)2% or (&gt;)72%</td>
</tr>
<tr>
<td>3(&lt;) and (\leq)14% or 89(&lt;) and (\leq)100%</td>
<td>2(&lt;) and (\leq)10% or 54(&lt;) and (\leq)72%</td>
</tr>
<tr>
<td>14(\leq) and (&lt;)27% or 59(&lt;) and (\leq)89%</td>
<td>10(&lt;) and (&lt;)20% or 36(&lt;) and (\leq)54%</td>
</tr>
<tr>
<td>27(\leq) and (\geq)59%</td>
<td>20(&lt;) and (\leq)36%</td>
</tr>
<tr>
<td>Native shrub cover(^B) (5): 1%</td>
<td>Native shrub cover(^B) (5): 9%</td>
</tr>
<tr>
<td>0 or (&gt;)4%</td>
<td>0 or (&gt;)26%</td>
</tr>
<tr>
<td>3(&lt;) and (&lt;)4%</td>
<td>0(&lt;) and (&lt;)3% or (&gt;)26%</td>
</tr>
<tr>
<td>2(&lt;) and (\leq)3%</td>
<td>3(&lt;) and (&lt;)5% or 13(&lt;) and (\leq)20%</td>
</tr>
<tr>
<td>0(&lt;) and (\leq)2%</td>
<td>5(&lt;) and (\leq)13%</td>
</tr>
<tr>
<td>Native cover(^B) other(^D) (10): 5%</td>
<td>Native cover(^B) other(^D) (10): 4%</td>
</tr>
<tr>
<td>0 or (\geq)16%</td>
<td>0 or (\geq)10%</td>
</tr>
<tr>
<td>0(&lt;) and (&lt;)1% or 11(\leq) and (\leq)16%</td>
<td>0(&lt;) and (&lt;)1% or 7(\leq) and (\leq)10%</td>
</tr>
<tr>
<td>1(\leq) and (&lt;)2% or 8(\leq) and (\leq)11%</td>
<td>1(&lt;) and (&lt;)2% or 5(&lt;) and (\leq)7%</td>
</tr>
<tr>
<td>2(&lt;) and (\leq)8%</td>
<td>2(&lt;) and (\leq)5%</td>
</tr>
<tr>
<td>Exotic plant cover(^B) (5)</td>
<td>Exotic plant cover(^B) (5)</td>
</tr>
<tr>
<td>(\leq)90%</td>
<td>(\geq)90%</td>
</tr>
<tr>
<td>(\geq)50-90%</td>
<td>(&lt;)50-90%</td>
</tr>
<tr>
<td>20(&lt;) and (\leq)50%</td>
<td>20(&lt;) and (\leq)50%</td>
</tr>
<tr>
<td>(\leq)20%</td>
<td>(\leq)20%</td>
</tr>
<tr>
<td>Hollow bearing trees (30): 14 trees ha(^{-1})</td>
<td>Hollow bearing trees (30): 60 trees ha(^{-1})</td>
</tr>
<tr>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>(&lt;)7 trees ha(^{-1})</td>
<td>(&lt;)30 trees ha(^{-1})</td>
</tr>
<tr>
<td>7(\leq) and (&lt;)14 trees ha(^{-1})</td>
<td>30(\leq) and (&lt;)60 trees ha(^{-1})</td>
</tr>
<tr>
<td>(\geq)14 trees ha(^{-1})</td>
<td>(\geq)60 trees ha(^{-1})</td>
</tr>
<tr>
<td>Regeneration of overstorey species(^E) (10)</td>
<td>Regeneration of overstorey species(^E) (10)</td>
</tr>
<tr>
<td>None</td>
<td>None</td>
</tr>
<tr>
<td>(&lt;)50%</td>
<td>(&lt;)50%</td>
</tr>
<tr>
<td>50(\leq) and (&lt;)100%</td>
<td>50(\leq) and (&lt;)100%</td>
</tr>
<tr>
<td>100%</td>
<td>100%</td>
</tr>
<tr>
<td>Length of fallen logs (5): 308 mha(^{-1})</td>
<td>Length of fallen logs (5): 643 mha(^{-1})</td>
</tr>
<tr>
<td>(&lt;)31 mha(^{-1})</td>
<td>(&lt;)64 mha(^{-1})</td>
</tr>
<tr>
<td>31(\leq) and (&lt;)154 mha(^{-1})</td>
<td>64(\leq) and (&lt;)322 mha(^{-1})</td>
</tr>
<tr>
<td>154(\leq) and (&lt;)308 mha(^{-1})</td>
<td>322(\leq) and (&lt;)643 mha(^{-1})</td>
</tr>
<tr>
<td>(\geq)308 mha(^{-1})</td>
<td>(\geq)643 mha(^{-1})</td>
</tr>
</tbody>
</table>
Figure 26: The use of broad classes to score attributes in the Vegetation Condition Score of Gibbons et al. (2004) masked differences between sites and awarded most sites with high scores. Demonstrated here for the native grass cover component of the Vegetation Condition Score: (a) the distribution of native grass cover amongst 16 woodland and 32 dry sclerophyll forest study sites; (b) the distribution of native grass cover scores for the same sites.
Chapter 8: Comparing the performance of the index to other indices

Figure 27: The Vegetation Condition Score (VCS$_{BM}$) component of the BioMetric Assessment Tool of Gibbons et al. (2004) had a distribution skewed towards high scores when applied to 32 dry sclerophyll forest study sites and 16 woodland study sites (a). It awarded similar high scores to groups of sites (■), whereas my proposed index differentiated between these particular sites (b).
8.4 Comparing the discrimination power of the different indices

Up to this point, my critique of indices has focused on comparing the performance of each index to the scoring and ranking of sites by my proposed index. While this approach was useful for highlighting the similarities and differences between my proposed index and the various indices, it does not provide an objective measure by which to assess the performance of all the indices, including my proposed one. To do this, I compared the discrimination power of the different indices. I defined the discrimination power of an index in terms of two parameters:

1. The average distance between sites on the index scale,
2. The percentage of sites with unique index values.

I used these parameters because they assessed discrimination at two different scales. The first parameter assessed the ability of an index to separate sites into distinct groups (large scale discrimination), and the second parameter assessed the ability of an index to identify individual sites within groups (fine scale discrimination).

When applied to my study sites, there were distinct differences in the discrimination power of the various indices (Figure 28). The Habitat Complexity scores of Newsome and Catling (1979) and Watson et al. (2001) had the lowest discrimination power. These indices tended to cluster sites at the low end of the index scale, and to score a high proportion of sites with the same value. I attributed the poor large scale discrimination of these indices to the scoring of cover on an absolute rather than relative scale, which meant that most sites scored low values. I attributed their poor fine scale discrimination to the use of broad classes to score attributes and to the limited number of attributes used in these indices. The marginally better performance of Newsome and Catling’s Habitat Complexity Score, compared to Watson et al.’s Habitat Complexity Score, was attributed to the inclusion of an attribute describing soil moisture, which varied between sites. Although strictly speaking this attribute relates to topographic position, rather than to stand structure.
Figure 28: There were clear differences in the discrimination power of my proposed index and five other indices. Discrimination power was assessed in terms of a) The average distance between sites on the index scale, b) The percentage of sites with unique index values. HCS\textsubscript{W} (Habitat Complexity Score, Watson et al., 2001); HCS\textsubscript{N} (Habitat Complexity Score, Newsome and Catling, 1979); VCS\textsubscript{BB} (Vegetation Condition Score, Oliver and Parkes, 2003); VCS\textsubscript{BM} (Vegetation Condition Score, Gibbons et al., 2004); SCS\textsubscript{HH} (Site Condition Score, Parkes et al., 2003)

The VCS\textsubscript{BM} of Gibbons et al. (2004) and the SCS\textsubscript{HH} of Parkes et al. (2003) had the best large scale discrimination of the various indices, I attributed this to the scoring of attributes on a relative rather than absolute scale and the inclusion of a wider range of attributes than the Habitat Complexity Score Indices. However, at a fine scale, their discrimination was relatively poor and I attributed this to their similar use of broad classes to score attributes.

The performance of the VCS\textsubscript{BB} of Oliver and Parkes (2003) was different to that of the SCS\textsubscript{HH}. This was unexpected because these two indices share some common attributes and scoring procedures. The improved fine scale discrimination of the VCS\textsubscript{BB} I attributed to the scoring of cover and richness as
separate attributes, in conjunction with multiple benchmarks to define the range of condition for these attributes.

Compared to the other five indices my proposed index performed well. At a large scale its discrimination was comparable to the VCS\textsubscript{BM} and SCS\textsubscript{HH}. This reflected the use of a comprehensive set of attributes in my index and their scoring relative to the range of values occurring in the two different vegetation communities. At a fine scale my index performed better than the other indices and discriminated between 100\% of sites. This demonstrated the effectiveness of using continuous functions rather than broad classes to score attributes.
Chapter 9: Conclusions

9.1 Introduction
In this final chapter I set out the main conclusions to be drawn from my study. These concern five key issues associated with quantifying stand structural complexity:

- Defining stand structural complexity;
- Identifying, scoring and combining structural attributes in an index;
- The practical application of this index;
- Applying the index to other vegetation types;
- Further research.

In the following sections I discuss the main findings for each of these issues in turn.

9.2 Defining stand structural complexity
I reviewed the literature concerning forest and woodland structure and found that stand structural complexity was not a clearly defined concept. To address this issue, I defined stand structural complexity as a combined measure of the number of different structural attributes present in a stand, and the relative abundance of each of these attributes. In this definition I included functional and compositional attributes under the umbrella of structural attributes because the division of attributes into three groupings was not a clear categorisation.

I attributed the range of stand structures present in an environment to different distributions of site resources amongst structural attributes, where simple structures often have high levels in a few attributes, with many attributes absent or present at very low levels, and complex structures often have medium to high levels in a wide range of attributes. This was analogous to approaches which have quantified diversity in terms of the abundance and richness of elements.

It was concluded that stand structural complexity is a relative, rather than absolute concept, because the potential levels of different structural attributes are bound within certain limits determined by the inherent characteristics of the site in question, and the biota of the particular community will have evolved to
reflect this range of variation. The presence of stands with naturally simple structures can therefore increase the diversity of habitats in the landscape, and so contribute to beta diversity. This was an important finding, because it implied that vegetation communities with naturally simple structures should have the potential to achieve high scores on an index of structural complexity.

9.3 Identifying, scoring and combining structural attributes in an index

I proposed and tested a methodology for developing an index of structural complexity. Using data from 144 plots in 48 study sites, located in woodland and dry sclerophyll forests, it was found that this methodology was an effective and objective basis for identifying, scoring and combining structural attributes in an index.

9.3.1 Identifying attributes

I established a comprehensive set of structural attributes as an objective starting point for developing the index. I demonstrated that the literature concerning fauna-habitat relationships and stand structural complexity provided a documentable basis for doing this. Thirteen core attributes were identified from this comprehensive set, by applying four clearly defined criteria to data collected from stands representing the range of condition, from modified to unmodified.

This approach was different to approaches used in other indices, in which relatively few authors provide quantitative evidence for selecting the attributes they use. In particular, Parkes et al. (2003) and Oliver and Parkes (2003), establish the attributes in their indices on the basis of expert opinion. This approach lacks transparency to those outside the group of experts, and can be biased by the makeup of the expert panel. While the use of expert opinion is justified in the absence of alternative knowledge or data, I demonstrated that there is a more robust option available.

Despite the different approach used to identify core attributes, there was partial agreement between my thirteen core attributes and those included in the
benchmarking indices of Parkes et al. (2003), Oliver and Parkes (2003) and Gibbons et al. (2004). Taken as a group, these indices contained attributes describing compositional diversity, coarse woody debris, regeneration, large trees and hollow trees – these were attributes that I also identified as core ones. However, unlike these indices, I quantified weeds indirectly through their effect on indigenous plant diversity, I included the contribution of non-indigenous species to vegetation cover and did not apply a discount to this contribution, I limited the direct assessment of regeneration to long-lived overstorey species, I used stand basal area as a surrogate for canopy cover, I quantified litter in terms of biomass (dry weight) rather than cover, and I included the additional attributes of quadratic mean diameter and the number of dead trees.

9.3.2 Scoring attributes

As a basis for constructing an index I rescaled the core attributes previously identified, as a score from 0-10. Continuous functions were used to do this, because this maintained a high correlation ($r > 0.97$, $p < 0.0001$) between attribute scores and the original attribute data. These functions were established by fitting regression equations to the quartile mid-points of the distribution of attribute values in study sites. This was an original approach to scoring, with most other indices using subjectively established classes to score attributes. In most cases, the boundaries used to define these classes were not related to any observed distribution of attribute values. I used field data to demonstrate that consequently these classes were often inappropriate for woodland and dry sclerophyll communities, because they masked differences between sites.

For each vegetation community, my scoring approach compared attributes to the range of values occurring in all stands, modified and largely unmodified, rather than to a benchmark range based on average levels in mature undisturbed stands (Parkes et al., 2003, Oliver and Parkes, 2003), or in stands relatively unmodified by post-European settlement (Gibbons et al., 2004). I concluded that these authors misapplied the concept of benchmarking because they benchmarked at the level of individual attributes, rather than at the level of structural complexity, which is the result of the interaction of these attributes.
Chapter 9: Conclusions

Characterising attributes in terms of a benchmark range or average level, ignores processes which underpin variation at the stand level, such as the increased development of some attributes at particular successional stages (Spies and Franklin, 1991; Noble and Slatyer, 1980), and the fact that attributes can respond differently to disturbance agents (Franklin et al., 2002; Purdie and Slatyer, 1976). It also produces a metric which is not particularly sensitive to the differences in attribute levels occurring between stands.

A more appropriate application of benchmarking would be at the overarching level of stand structural complexity, using a metric such as the index developed in this thesis. These benchmarks could, for example, reflect observed levels of structural complexity in unmodified natural stands at different successional stages, or thresholds for structural complexity at which a wide range of biota are present. Such benchmarks or thresholds could define useful goals for guiding management, and this is discussed further in section 9.6.1.

9.3.3 Combining and weighting attributes

I agreed with Parkes et al. (2003), Oliver and Parkes (2003), Gibbons et al. (2004), Watson et al. (2001) and Newsome and Catling (1979), that a simple additive index was an efficient framework for combining the rescaled core attributes in an index of structural complexity. In this form, my index was straightforward to apply, approximately normally distributed amongst study sites, and differentiated between study sites better than the five prominent indices I tested. However, it should be noted that resource and time constraints precluded the use of a new and independent data set for this testing, so that the superior performance of my index should be interpreted cautiously.

Unlike most other indices, I used sensitivity analysis to assess the effect of weighting attributes in my proposed index. This analysis indicated that my index was not sensitive to attribute weightings, and on this basis attributes carried equal weight in the final index. The robust performance of the unweighted index was attributed to the inclusion of thirteen attributes, each of which was approximately normally distributed and able to differentiate between study sites. Consequently any substantial change in the weighting of a particular attribute
was buffered by information contained in the other attributes. This approach contrasted with Parkes et al. (2003), Oliver and Parkes (2003) and Gibbons et al. (2004), who applied unequal weightings to attributes. In the case of Parkes et al. (2003), and Oliver and Parkes (2003) the size of these weightings appeared a matter of subjective judgement, which reflected the relative value the authors placed on each attribute, rather than any quantitative assessment of the effect weighting had on the performance of the particular index.

9.4 The practical application of the index

I demonstrated the practical application of my index in a user-friendly spreadsheet, designed to allow landowners and managers to assess the condition of their vegetation, and to identify management options. This spreadsheet calculated an index score from field data, and then used this score to rank the site relative to a set of reference sites, which were indicative of the range of condition in the study area. Although this ranking did not account for the spatial relationship between sites and its impacts on biodiversity, it nevertheless added a regional context to the operation of my index, and is a potentially useful tool for identifying sites of high conservation value, or for identifying sites where management actions have maintained vegetation quality. This was a different approach to the application of benchmarking by Parkes et al. (2003), Oliver and Parkes (2003) and Gibbons et al. (2004), because it compared each site to the measured range of condition in a region, rather than to an assumed ideal level of condition.

The spreadsheet incorporated the option of calculating an index score using a subset of attributes, and provided a measure of the uncertainty associated with this score, so that the cost of using a subset of variables was made explicit. This feature allows the index to be applied to existing data sets, which may not include data for all 13 attributes. It also allows managers to rationalise the effort they put into data collection, because some attributes are more time consuming to quantify than others. This was an original approach, not included in other indices.

The spreadsheet also provided information on the status of individual attribute
levels, by classifying them as high (in the top third of reference site values), average (in the middle third of reference site values), or low (in the bottom third of reference site values). The use of these three classes was considered informative, because it provided a simple scale for highlighting the current status of each attribute and for setting future management options. In this approach, attributes were scored initially as continuous variables, which were converted to classes at the end of the analysis. This maximised the information captured in the index, and ensured that class boundaries related to the actual distribution of data amongst reference sites. This approach is different to the indices of Parkes et al. (2003), Oliver and Parkes (2003), Gibbons et al. (2004), Watson et al. (2001) and Newsome and Catling (1979), in which attributes are scored on the basis of predetermined classes.

9.5 Applying the index to other vegetation types
The performance of my index demonstrated the importance of scoring attributes relative to the range of values occurring in stands of a comparable vegetation community. A new set of reference sites should therefore be established for each vegetation type the index is applied to. Data from these sites would then provide the basis for developing the necessary functions for scoring attributes in the new vegetation type. In the absence of these data, the application of the index should be limited to communities that are similar in structure to the woodland and dry sclerophyll forest communities quantified in this study.

The index can also be applied to non-indigenous vegetation, such as a plantation, which occurs on a site formerly occupied by either a woodland or dry sclerophyll forest community. To calculate an index value for the non-indigenous vegetation, attributes would be quantified as for an indigenous site, except that all perennial species and lifeforms are counted, indigenous or otherwise. This index value is then used to rank the non-indigenous site relative to the appropriate reference sites (woodland or dry sclerophyll forest). In this approach, attribute levels in the non-indigenous community are compared to the range of values set by the indigenous reference sites. This approach makes no assumptions about the worth of non-indigenous species, but simply compares the structural complexity achieved by the non-indigenous vegetation to the
range of structural complexity associated with the original indigenous vegetation.

9.6 Further research

As a closing comment, there are two areas of further research, which can build on the results of this thesis. The first is the identification of structural complexity thresholds for use by landowners, and the second is the use of remote sensing techniques to apply the index over large areas, and to integrate the index with an assessment of landscape scale attributes.

9.6.1 Thresholds

For many landowners, a key concern is whether their current management actions are degrading or maintaining vegetation quality on their property. To answer this question landowners require a threshold value for structural complexity, which will allow them to delineate degraded from intact vegetation.

The use of three classes (high, average, low) in the spreadsheet application of my index partly addressed this issue, by providing a simple scale highlighting the status of each attribute, relative to the range of levels in reference sites. These classes were subjectively established by dividing the distribution of attribute values in reference sites into three equal groups. Additional research could provide a more objective basis for establishing these classes or other thresholds. For example, monitoring the presence and abundance of different fauna in sites representing a range of index scores could establish a threshold value for structural complexity, below which many faunal species are absent. This research would also need to consider landscape variables such as patch size, and connectivity to other patches, because these impact on the presence and abundance of different fauna.

Such research would be in line with Lambeck’s (1997) focal species approach, which Watson et al. (2001) have demonstrated as an effective method for establishing structural complexity thresholds for woodland bird species. However the index developed in this thesis would provide a more robust measure of structural complexity than was available for that study.
9.6.2 Remote sensing

To apply the index to a large area such as a catchment, it will be necessary to develop techniques to remotely sense index values, because collecting field data for all patches in the landscape is impractical. This would also allow the index to be linked to metrics describing landscape attributes likely to affect the biodiversity value and long term viability of a patch, such as patch size, shape, connectivity to other patches, and area of adjacent remnant vegetation (Mahiny, 2004, McIntyre et al. 2002).

There is some research already addressing this issue. For example, Simpson (2004) is developing regression models, for predicting structural attributes of dry sclerophyll forest using spectral and spatial variables derived from Landsat imagery. Early results are promising, with Simpson (2004) reporting a significant linear relationship ($r^2 = 0.64$, $p<0.0001$) between spectral and spatial variables and index scores for my study sites.
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Recher, H. F., Majer, J. D. and Ganesh, S. (1996) Eucalypts, arthropods and...


Rehwinkle, R. (2002) pers. comm. NSW NPWS database of high quality remnant vegetation. c/o NSW NPWS, Queanbeyan, NSW.


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Appendix 1

Fieldwork proforma sheets for the collection data as described in Chapter 5.
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Vegetation cover

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Estimated cover in 10mx10m sections of 20mx20m quadrat (1m²=1% cover)

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Canopy cover - 0 (foliage absent), 1 (dense foliage), 0.5 (diffuse foliage)

<table>
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<table>
<thead>
<tr>
<th>Tree height</th>
<th>Tree 1</th>
<th>Tree 2</th>
<th>Tree 3</th>
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Estimated in 10mx10m quadrats

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<th>49</th>
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T tree, SL shrub <0.5m, ST shrub >0.5m, GT grass tussock, GNT grass non-tussock, SR sedge rush, FB forb, FN fern, V vine, RGL regeneration <2m, RGT regeneration 2-4m, X xanthorrhoea
### Logs and litter

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<thead>
<tr>
<th>Site ID</th>
<th>Plot#</th>
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<table>
<thead>
<tr>
<th>Diameter (cm)</th>
<th>Length (m)</th>
<th>Decay class (1-3)</th>
<th>Diameter (cm)</th>
<th>Length (m)</th>
<th>Decay class (1-3)</th>
</tr>
</thead>
<tbody>
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**Litter**

<table>
<thead>
<tr>
<th>Bag</th>
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</thead>
<tbody>
<tr>
<td>Bag 1</td>
<td>5</td>
</tr>
<tr>
<td>Bag 2</td>
<td>15</td>
</tr>
<tr>
<td>Bag 3</td>
<td>25</td>
</tr>
<tr>
<td>Bag 4</td>
<td>35</td>
</tr>
<tr>
<td>Bag 5</td>
<td>45</td>
</tr>
</tbody>
</table>

**Key to decay classes**

1 = solid when kicked lacks cracks, cavities or hollow pipe
2 = solid when kicked contains cracks, cavities or hollow pipe
3 = gives or sloughs when kicked
Appendix 2

Structural variables quantified using the plot based measurements described in Chapter 5

<table>
<thead>
<tr>
<th>Structural attributes</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Standard deviation of basal area of live trees</td>
<td>Quadratic mean dbh of live trees live</td>
</tr>
<tr>
<td>Standard deviation of quadratic mean dbh</td>
<td>Coefficient of variation of dbh live</td>
</tr>
<tr>
<td>Number of live stems per ha</td>
<td>Number of live stems per ha with dbh &gt; 40</td>
</tr>
<tr>
<td>Number of live stems per ha with dbh &gt; 50</td>
<td>Number of live stems per ha with dbh &gt; 60</td>
</tr>
<tr>
<td>Number of live stems per ha with dbh &gt; 80</td>
<td>Number of regenerating stems per ha</td>
</tr>
<tr>
<td>Basal area of dead trees per ha</td>
<td>Quadratic mean dbh of dead trees</td>
</tr>
<tr>
<td>Coefficient of variation of dbh dead</td>
<td>Number of dead trees per ha</td>
</tr>
<tr>
<td>Number of stumps per ha</td>
<td>Quadratic mean dbh of stumps</td>
</tr>
<tr>
<td>Basal area of stumps</td>
<td>Basal area of trees with mistletoe</td>
</tr>
<tr>
<td>Number of hollow trees per ha</td>
<td>Number of trees per ha with small hollows</td>
</tr>
<tr>
<td>Number of trees per ha with medium hollows</td>
<td>Number of trees per ha with large hollows</td>
</tr>
<tr>
<td>Hollow Diversity Index</td>
<td>% cover of cryptogams</td>
</tr>
<tr>
<td>% cover of logs</td>
<td>% cover of rocks</td>
</tr>
<tr>
<td>% cover of bare ground</td>
<td>% cover of vegetation 0 - 0.5m</td>
</tr>
<tr>
<td>% cover of vegetation &gt;0.5</td>
<td>% cover of vegetation 0.5 - 2m</td>
</tr>
<tr>
<td>% cover of vegetation 2 - 4m</td>
<td>% cover of vegetation 4 - 6m</td>
</tr>
<tr>
<td>% canopy cover of all species</td>
<td>% canopy cover of exotic species</td>
</tr>
<tr>
<td>Coefficient of variation of % canopy cover</td>
<td>Canopy health</td>
</tr>
<tr>
<td>Site Height</td>
<td>% cover of native shrubs</td>
</tr>
<tr>
<td>% cover native shrub &gt;0.5m</td>
<td>% cover exotic shrub &gt;0.5m</td>
</tr>
<tr>
<td>% cover native shrub &lt;0.5m</td>
<td>% cover exotic shrub &lt;0.5m</td>
</tr>
<tr>
<td>% cover native tussock grasses</td>
<td>% cover exotic tussock grasses</td>
</tr>
<tr>
<td>% cover native sedges and rushes</td>
<td>% cover exotic sedges and rushes</td>
</tr>
<tr>
<td>% cover native ferns</td>
<td>% cover exotic ferns</td>
</tr>
<tr>
<td>% cover native vines</td>
<td>% cover exotic vines</td>
</tr>
<tr>
<td>% cover of all exotic species</td>
<td>% cover of overstorey regeneration &lt;2m</td>
</tr>
<tr>
<td>% cover of overstorey regeneration &gt;2m</td>
<td>% total cover of overstorey regeneration</td>
</tr>
<tr>
<td>Perennial species richness</td>
<td>Lifeform richness</td>
</tr>
<tr>
<td>Shrub species richness</td>
<td>Compositional diversity using Shannon Weiner Index</td>
</tr>
<tr>
<td>% cover of litter</td>
<td>Coefficient of variation of % cover of litter</td>
</tr>
<tr>
<td>Litter dry weight</td>
<td>Coefficient of variation of litter dry weight</td>
</tr>
<tr>
<td>Decay class1 log volume per ha</td>
<td>Decay class1 log length per ha</td>
</tr>
<tr>
<td>Decay class1 large log length per ha</td>
<td>Decay class2 log volume per ha</td>
</tr>
<tr>
<td>Decay class2 log length per ha</td>
<td>Decay class2 large log length per ha</td>
</tr>
<tr>
<td>Decay class3 log volume per ha</td>
<td>Decay class3 log length per ha</td>
</tr>
<tr>
<td>Decay class3 large log length per ha</td>
<td>Total log volume per ha</td>
</tr>
<tr>
<td>Total log length per ha</td>
<td>Total large log length per ha</td>
</tr>
<tr>
<td>Coarse Woody Debris Diversity Index</td>
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</table>