Management Principles and Strategies to Guide Biodiversity Conservation in Private Native Forests

A checklist for forest and wildlife managers

RIRDC Publication No. 09/032
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by D. B Lindenmayer

March 2009

RIRDC Publication No 09/032
RIRDC Project No CAR-2A
Foreword

Conservation of biodiversity is a fundamental criterion underpinning ecological sustainability in forest management including in private native forests. This report outlines a series of general management principles and a checklist of strategies for management of private native forests. The checklist is ordered hierarchically from large scale to individual tree based approaches for integrating biodiversity conservation and wood production.

The focus is on a set of general principles rather than specific prescriptions because precise prescriptions vary significantly according to location, landscape, biotic assemblages and management objectives. Nevertheless the general principles and hierarchically structured checklist provide a benchmark against which codes of practice and forestry standards (including those intended for certification) can be assessed.

This project is part of a suite of private native forestry projects funded by the Natural Heritage Trust through the Joint Venture Agroforestry Program (JVAP). JVAP is supported by three R&D Corporations – Rural Industries Research and Development Corporation (RIRDC), Land & Water Australia (L&WA), and Forest and Wood Products Research and Development Corporation (FWPRDC). The Murray-Darling Basin Commission (MDBC) also contributed to this project. The R&D Corporations are funded principally by the Australian Government. State and Australian Governments contribute funds to the MDBC.

This report is an addition to RIRDC’s diverse range of over 1800 research publications. It forms part of our Agroforestry and Farm Forestry R&D program, which aims to integrate sustainable and productive agroforestry within Australian farming systems. The JVAP, under this program, is managed by RIRDC.

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Peter O’Brien
Managing Director
Rural Industries Research and Development Corporation
Acknowledgments

This report arose as a direct result of a collaborative book-writing project with Professor Jerry Franklin. This project was supported by NHT through the Joint Venture Agroforestry Program, managed by Dr. Rosemary Lott. Guidance from David Thompson, Phil Norman, Warwick Ragg and Dr. Rosemary Lott was greatly appreciated. Insightful comments on the manuscript by Alex Jay, David Thompson and Warwick Ragg were greatly appreciated.
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Executive Summary

What the report is about

In this report, a series of ecological principles is outlined to guide how forestry on private land might best achieve biodiversity conservation. A checklist of on-ground management strategies aimed at operationalising these principles is described. A discussion of the meaning of ecologically sustainable forest management, together with an outline of problems associated with biodiversity conservation “short-cuts” is given to provide suitable context to the description of the principles and associated checklist. Importantly, the focus of this paper is on existing areas of native forest rather than exotic plantations (including eucalypt plantations) or farm forests that have been established on former grazing lands.

Who is the report targeted at?

This report addresses issues that should be of interest and importance to forest managers and those involved in developing forest management plans both on public lands and in private native forests.

Background

Private native forests comprise a significant proportion of the cover of native forest in many Australian jurisdictions. As such, their sustainable management is a critical issue for the maintenance of forest productivity and forest biodiversity.

Many kinds of indicators and criteria have been proposed as measures to assess the sustainable management of forests (both public and private) but the scientific validity of many of these surrogates remain highly uncertain, particularly in terms of their validity for assessing management practices that aim to integrate production with the maintenance of biodiversity. Problems arise because of the wide range of responses of different elements of the biota to disturbances (such as logging) which often lead to species-specific, site-specific, landscape-species, forest-type-specific, and region-specific outcomes that are not readily generalized to other species, sites, landscapes, forest types and regions. In this document, problems with ecological “shortcuts” such as indicator species, focal species, ecological thresholds and landscape indices are discussed. Such problems mean their use as indicators or surrogates for sustainability in forest management are not recommended, including on private land. An alternative approach is to develop a set of guiding principles for biodiversity conservation that is broadly applicable to any forested area – those on public land and on private land.

Aims/objectives

This report outlines a series of general management principles and a checklist of strategies for management of private native forests. The focus is on general principles rather than specific prescriptions because the latter vary significantly according to location, landscape, biotic assemblages and management objectives. Nevertheless the general principles and hierarchically structured checklist provide a benchmark against which codes of practice and forestry standards (including those intended for certification) can be assessed.

Methods used

The overarching goal for conserving forest biodiversity is the maintenance of suitable habitat. Five general principles can help meet this goal:

- the maintenance of connectivity
- the maintenance of landscape heterogeneity
• the maintenance of stand structural complexity
• the maintenance of aquatic ecosystem integrity
• the use of natural disturbance regimes to guide human disturbance regimes.

Allied with these guiding principles, a checklist of measures for forest biodiversity conservation is presented, reflecting the multi-scaled nature of conservation approaches on forested land.

The general principles and checklist provide a benchmark against which new codes of practice might be developed or the efficacy of existing codes assessed and deficiencies in them subsequently addressed. The checklist also provides a basis against which performance in forest management practices can be gauged relative to guidelines like those presented in the Australian Forestry Standard. Notably, such codes and forestry standards typically do not take into account issues outlined in the checklist like the importance of natural disturbance refugia, and the potential problems created by multiple cumulative impacts (e.g. the combination of grazing, logging and fire).

Results/key findings

The checklist can be applied either to public forest or private native forest. While the checklist is simple and intuitive, its elements have rarely been presented as a hierarchical list as outlined in this report. The checklist outlined in this report embodies a continuum of management approaches arranged in approximate hierarchical order progressing from very large-spatial-scale strategies (large ecological reserves) through mid-spatial-scale or landscape-level strategies to stand-level silvicultural approaches (e.g. stand structural retention).

Forest managers often struggle to deal with the complexity that subsequently results from stand-, landscape- and region-specific solutions to integrating conservation and wood production. However, consideration of the general ecological principles, together with consideration of the elements of the checklist should assist the development of more credible forest conservation plans both on public lands and in private native forests. The checklist includes:

• the establishment of large ecological reserves (which usually will not apply to private native forests)

• off-reserve conservation measures at the landscape level such as:
  − protected areas within production forests
  − establishment of riparian buffers
  − appropriate location of road networks
  − spatial and temporal arrangement of harvest units
  − appropriate fire management practices.

• off-reserve conservation measures at the stand level such as:
  − retention of key elements of stand structural complexity at the time of harvest such as large trees with hollows (plus recruits), understorey thickets and large fallen logs
  − longer rotation times (coupled with structural retention at harvest)
  − altered silvicultural systems (including novel approaches to retention and the application of modified thinning regimes)
— appropriate fire management practices.

**Implications for relevant stakeholders**

Specific prescriptions for on-the-ground implementation of the general principles outlined in the checklist are not provided in this report. This is because differences in the biota between different forest types and regions, variations in the responses to such biota to disturbance, and a myriad of other factors mean there will be no universal forest biodiversity conservation ‘recipes’ that can be applied uniformly and uncritically to all regions, landscapes and stands. Rather, the best and/or most appropriate strategies will vary according to particular forest management, conservation and other objectives within particular areas. This makes it imperative for forest managers (on any land – including private native forests) to be clear and explicit about the objectives of their management actions. Unfortunately, this is rarely done in a formal way that lends itself for assessment against sustainability criteria.

**Recommendations**

Although the general ecological principles and associated checklist are intuitive, there are limited data to determine the effectiveness of many particular on-the-ground management actions. Therefore, considerable new effort is needed to adopt adaptive management “natural experiments” and monitoring to:

1. better identify the impacts of logging operations and other kinds of management of biodiversity, and

2. quantify the efficacy of impact mitigate strategies and ways to improve them where necessary.

It is argued that a commitment to true adaptive management that involves rigorous monitoring and a commitment to management change when negative impacts are identified should be considered to be a key element of ecologically sustainable forest management. Unfortunately, the record on forest monitoring (and particularly on biodiversity) remains extremely poor both on public native forests and on private forested land in all Australian jurisdictions.
Introduction

Forests are some of the most species-rich ecosystems on earth and support about 65% of the world’s terrestrial taxa (World Commission on Forests and Sustainable Development, 1999). They are the most species-rich environments on earth for many groups ranging from birds (Gill, 1995) and invertebrates (Erwin, 1982; Majer et al., 1994) to microbes (Torsvik et al., 1990; Crozier et al., 1999). Conserving forest biodiversity is therefore a critical task (Hunter, 1999) and has rightly become a key component of many national and international agreements associated with forest management (e.g. Commonwealth of Australia, 1998, 2001; Arborvitae, 1995; Montreal Process Liaison Office, 2000; Food and Agriculture Organisation of the United Nations, 2001).

Most efforts in sustaining forest biodiversity, both overseas and in Australia have focused on setting aside reserves and other kinds of protected areas. Indeed, reserves are a critical part of any credible strategy for conserving forest biodiversity (Norton, 1999). However, it is clear that reserves alone will be insufficient to adequately conserve forest biodiversity (Hale and Lamb, 1997; Sugal, 1997; Lindenmayer and Franklin, 2002), not the least because approximately 92% of the world’s forests are outside the formal networks of protected areas (Commonwealth of Australia, 1999). Levels of forest reservation on public lands are significantly higher than this for many forest types in Australia (Commonwealth of Australia, 2004). Nevertheless, forests outside the reserve system (including those on private land) have a significant role to play, not only in the conservation of forest biodiversity, but also in the maintenance of other key aspects of sustainability (such as the maintenance of soil productivity and water quality).

1.1. The conservation significance of private native forests

In Australia, private native forests are important with respect to biodiversity conservation for several key reasons. Three of these are:

- the area of private native forest (defined as forest not managed by government agencies) is large in some jurisdictions (Commonwealth of Australia, 2004) and will therefore support large (and hence significant) populations of many forest-dependent species.

- despite recent additions to the protected area network in Australia, some forest types remain poorly protected in the reserve system. In addition, the formal reserve system remains biased toward steep terrain on poor soils (Pressey, 1995; Pressey et al., 1996; Lindenmayer and Franklin, 2002). Such low productivity areas can have limited conservation value for some elements of the biota (Braithwaite, 1984; Braithwaite et al., 1993). Conversely, in many jurisdictions, forest on private lands is typically on the most productive areas and has considerable significance for some species (Braithwaite, 2004). Careful management is needed to best integrate forestry operations and conservation values in such areas of private native forest.

- some species move over large distances and respond to the spatial arrangement of suitable habitat across land tenures (Date et al., 1996; Price et al., 1999) such as state forests and private native forests (Brereton, 1997). The maintenance of habitat on private native forest will be critical for the persistence of these species. An example is the Swift Parrot (Lathamus discolor) in Tasmania. Less than 5% of the nesting habitat of the species occurs in dedicated conservation reserves with the rest on private and publicly-owned production forests (Brereton, 1997). The long-term conservation of the Swift Parrot depends almost entirely on management actions outside reserves (Brown and Hickey, 1990; Swift Parrot Recovery Team, 2000; Lindenmayer et al., 2003).

Given the importance of private native forests for biodiversity conservation in Australia, in this report, a series of ecological principles is outlined to guide how forestry on private land might best achieve biodiversity conservation. A checklist of on-ground management strategies aimed at operationalising
these principles is described. A discussion of the meaning of ecologically sustainable forest management, together with an outline of problems associated with biodiversity conservation “shortcuts” is given to provide suitable context to the description of the principles and associated checklist. Importantly, the focus of this paper is on existing areas of native forest rather than exotic plantations (including eucalypt plantations) or farm forests that have been established on former grazing lands. Other work tackles issues associated with biodiversity conservation in plantations (e.g. Lindenmayer and Hobbs, 2004) and farm forests (Salt et al., 2004). It is re-emphasized that the focus of this report is on sustaining populations of native biota, although it is well recognized that many other factors are a fundamental part of sustainability such as the maintenance of soil quality, productivity etc.
2. Ecologically sustainable forest management and the conservation of biodiversity

For the purposes of this paper, ecologically sustainable forestry can be broadly considered as (after Lindenmayer and Recher, 1998):

“…perpetuating ecosystem integrity while continuing to provide wood and non-wood values; where ecosystem integrity means the maintenance of forest structure, species composition, and the rate of ecological processes and functions with the bounds of normal disturbance regimes.”

In the context of this definition, it is important to determine appropriate baselines and ranges of variability for natural disturbance regimes against which human disturbance regimes can be compared (Hunter, 1993; Lindenmayer and Franklin, 2002). Achieving ecological sustainable forestry in Australia’s forests (both on public and private land) will not be easy, not the least because eucalypt forests are extraordinarily diverse and rich in species (Common and Norton, 1992; Recher, 1996). The conservation of biological diversity is clearly part of perpetuating ecosystem integrity (as highlighted in the above definition). Hence, it is one of the goals of ecologically sustainable forestry, although as stated earlier, the concept of sustainability encompasses much more than biodiversity conservation alone. There are numerous definitions of biodiversity (Delong, 1996 and Bunnell, 1998 reviewed approximately 90 interpretations of the concept!), but in this report, biodiversity is considered to encompass (after Lindenmayer and Burgman, 2005):

“….genes, individuals, demes, populations, metapopulations, species, communities, ecosystems, and the interactions between these entities.”

This definition stresses both the numbers of entities (genes, species etc) and the differences within and between those entities (see Gaston and Spicer, 2004). Given this complexity and the inadequate description of biodiversity currently available (e.g. Erwin 1982; Torsvik et al.1990; Majer et al.1994), it is difficult to judge whether or not forests are being managed in an ecologically sustainable way (Lindenmayer et al., 2000). Indeed, there are no cases anywhere in the world where ecologically sustainable forestry practices have been demonstrated unequivocally (Botkin and Talbot 1992; Bunnell et al., 2003; Lindenmayer and Franklin 2003). Moreover, the concept of sustainability per se is a complex and dynamic one. This is because ecological sustainability is an overall direction in conservation and forest management. There will be many transitions in the overall direction of ecological sustainability in forest management, and not all of the movement will necessarily be forward (Lindenmayer and Franklin, 2003).

Despite the problems of: (1) defining biodiversity, (2) determining what is (and is not) ecologically sustainable forestry, and, (3) the sheer impossibility of measuring and monitoring the impacts on all species of various management practices (Lindenmayer et al., 2000), an array of international and national initiatives (e.g. Arborvitae 1995) have sought to develop criteria and indicators of sustainability in forests. Many of these attributes and measures are aimed at serving as surrogates for biodiversity and, in turn, have been proposed for use in assessing the success or failure of management practices to sustain biodiversity. Some kinds of indicators of sustainability will be essential because of the sheer impossibility of measuring and monitoring everything (Lindenmayer and Burgman, 2005). However, as briefly outlined in the following section, the scientific validity of most indicators of biodiversity is poor. The set of guiding principles and accompanying checklist proposed later in this paper have been developed, in part, because of problems with concepts like the indicator species approach as discussed below.
3. Background – “short-cut” methods that attempt to promote ecologically sustainable forest management practices

The ecological and forest management literatures contain numerous examples of “short-cuts” that have aimed to promote biodiversity conservation as part of the sustainable management of natural resources – including native forests. The short-cuts examined briefly below are indicator species, focal species, indices of forest landscape fragmentation, and thresholds in levels of native vegetation cover. There are considerable problems with all of them. The aim of the remainder of this section is to briefly outline these problems and highlight the value of an alternative approach based on a set of general principles that can then be matched with a checklist of multi-scaled management strategies.

3.1. Indicator species

Landres et al. (1988, p. 317) defined an indicator species as:

‘an organism whose characteristics (e.g. presence or absence, population density, dispersion, reproductive success) are used as an index of attributes too difficult, inconvenient, or expensive to measure for other species or environmental conditions of interest.’

The indicator species approach has been widely proposed for use as part of ecologically sustainable forest management (Lindenmayer et al., 2000) and many different groups have been suggested as indicators including (among others): microbes, bryophytes, ants, butterflies, amphibians, small mammals, arboreal marsupials, large wide-ranging carnivores, and birds (reviewed by Lindenmayer and Burgman, 2005). Problems with the indicator species concept have been reviewed by Landres et al. (1988), Temple and Wiens (1989), Niemi et al. (1997), Simberloff (1998) and Lindenmayer et al. (2000). An extensive appraisal is beyond the scope of this report. However, the chief difficulties include:

- a general lack of understanding of causal relationships between indicator species and the entities they are assumed to indicate (Lindenmayer and Franklin, 2002). Indeed, the causal relationships between disturbance (or other forms of perturbation) and species response have not been established for any taxon recommended as an ‘indicator species’ (Lindenmayer et al., 2000)
- the major variation between species (even closely related ones) response to change, including among members of the same guild or other kinds of assemblages
- the insensitivity of some species to change
- the lack of knowledge to guide the selection of a given indicator species. For example, although many workers have contended that particular taxa are indicator species (e.g. Davey, 1989; Johnston, 1994; Hill, 1995), the entities they are supposed to indicate often are not explicitly stated (Lindenmayer and Cunningham, 1997).

In summary, while the notion of indicator species is a potentially useful one, much more work needs to be done to confirm the relationships between species chosen as indicators and environmental change (Simberloff, 1999; Kavanagh et al., 2004).
3.2. Focal species

The focal species concept is one closely allied with that of the indicator species approach. Focal species are the ones most influenced by threatening processes; they might be the most area-sensitive, dispersal-limited, resource-limited, and ecological-process-limited taxa in a landscape (Lambeck, 1999). The idea is to manage a landscape for a suite of focal species, each of which is thought to be sensitive to a particular threatening process. Lambeck (1997) claimed that “because the most demanding species are selected, a landscape designed and managed to meet their needs will encompass the needs of all other species.”

Lindenmayer et al. (2002a) and Lindenmayer and Fischer (2003) noted there were some serious flaws in the focal species approach. First, the underlying theoretical basis of the focal-species approach is problematic. As part of a taxon-based surrogate scheme, a suite of focal species is presumed to act collectively as a surrogate for other elements of the biota. However, taxon-based surrogate schemes have had limited success everywhere they have been applied (see section above on indicator species). On this basis, a landscape managed for a given set of focal species may not meet the requirements of the remaining biota. Second, the focal species approach may be unsuitable for practical implementation, primarily because of the lack of data to guide the selection of a set of focal species in the majority of landscapes. It seems highly unlikely that the focal species approach will survive rigorous scientific scrutiny because, for example, some of the fundamental principles of ecology (particularly the basis for species co-existence) must be breached for it to work (Lindenmayer et al., 2000). Perhaps the real success of the focal species approach will be its ability to act as a “social hook” to motivate people, communities and governments to seriously begin to tackle the difficult process of landscape management (Lindenmayer and Fischer, 2003).

3.3 Indices of forest fragmentation

Often, spatial patterns of vegetation cover created by landscape alteration are linked with the responses of particular elements of biodiversity. Spatial processes may be quantified by measuring the spatial patterns they create, particularly ‘patchiness’ and the size, shape, composition, juxtaposition and arrangement of landscape units (e.g. vegetation patches) (Wegner, 1994; Forman, 1995; Smith, 2000; Lindenmayer et al., 2002b; McAlpine et al., 2002a). Metrics to characterize such patterns are termed landscape indices and they are used widely. Indeed, indices of forest fragmentation are now a part of the Montreal Process criteria and indicators of sustainable forest management (Commonwealth of Australia, 1998; Tickle et al., 1998).

Landscape indices may be useful to help establish whether a pattern has changed (Wegner, 1994; Smith, 2000; McAlpine et al., 2002). They also may be valuable for linking spatial patterns of landscape cover to species responses. However, when using landscape indices to describe landscapes and guide management actions, it is important to be clear about the reason for using them and the inherent limitations of using such an approach. Indeed, Cale and Hobbs (1994) and Lindenmayer et al. (2002b) appraised and criticized the use of landscape indices. In particular, they noted:

- the methods used to develop indices often are not provided
- it is not always clear what to do with landscape indices when they are generated and they are often not well linked to appropriate aspects of land management
- few landscape indices are used consistently in different investigations
- landscape indices fail to account for factors such as the vertical heterogeneity of vegetation, known to be important for many elements of the biota such as birds and bats (e.g. Gilmore, 1985; Brown et al., 1997; Brokaw and Lent, 1999)
• many indices provide sophisticated ways of highlighting intuitively obvious landscape patterns and have led to few new insights

• many landscape indices are highly correlated (providing similar sorts of information about a landscape). This creates problems both for their use in statistical analyses and in their practical application on the ground.

• many studies generate large numbers of metrics and hence many more than the degrees of freedom available to facilitate robust statistical analyses

• each species responds differently to the same spatial scale of landscape change and human disturbance (e.g. Davies and Margules, 1998; Villard et al., 1999). Hence, no single measure adequately reflects change for all biota. In addition, landscape indices will fail when species responses and perceptions of a landscape are different from the way humans characterize and map that landscape (Lindenmayer et al., 2002c)

• most indices provide an instantaneous static measure, whereas temporal dynamics may be important, such as the length of isolation of vegetation remnants (Loyn, 1987; Bennett, 1990; Gascon et al., 1999) or the time since the last timber harvesting event (Fahrig, 1992)

• values for indices are often scale-dependent, making it difficult to compare results from different landscapes and spatial scales. In addition, the scale of a species’ movement was often not well linked to the scale at which landscape indices were generated.

In relation to the final point listed above, Cale and Hobbs (1994) stated that:

“Indices of [landscape] diversity can indicate that differences in heterogeneity exist when no actual change in habitat has occurred from the organism’s point of view, or (they) can fail to detect important changes in habitat. The principal reason for this is the problem of matching the scale of measurement with the scale at which organisms perceive the environment.”

3.4. Thresholds of vegetation cover

With and King (1999) defined thresholds as abrupt, non-linear changes that occur in some measure (such as the rate of loss of species) across a small amount of habitat loss. It has been hypothesized that where a threshold response occurs, below a critical amount of habitat cover, the loss of species and populations is greater than can be predicted from a linear relationship with habitat cover alone (Rolstad and Wegge, 1987; Andrén, 1994, 1999; Enoksson et al., 1995; With and Crist, 1995; With and King, 1999). The search for threshold responses for biota has become a topic of increasing interest in biodiversity-related research within human-modified landscapes (e.g. McAlpine et al., 2002b; Radford and Bennett, 2004) including those targeted for forest management (Andrén, 1994, 1999). If ecological thresholds are a widespread phenomenon in human-modified landscapes, there could be a valuable tool for use in landscape management such as planning the extent and spatial arrangement of use of natural resources.

A recent series of papers has examined issues on ecological thresholds (e.g. Lindenmayer and Luck, 2005), and it seems distinctly possible that threshold responses for aggregate measures such as species richness may not exist in some ecosystems. For example, a major investigation underway on an array of forest-dependent species in western Canada has shown the vast majority of species richness relationships and species-specific response relationships were smooth or curvilinear and few, if any, displayed characteristics of threshold functions (Bunnell et al., 2003). Similarly, Lindenmayer et al. (2005) was unable to detect threshold responses for native vegetation cover and bird and reptile diversity in the radiata pine (Pinus radiata) plantation system at Tumut in southern New South Wales. In addition, in cases where thresholds might exist, they will depend on the landscape in question (a forest-forest or forest-agriculture system), the assemblages or particular species of interest, and the
ecological processes in question (the extent of tree health or the extent of landscape-wide clearing). On this basis, it is unlikely there will be generic rules for critical change points or threshold levels of vegetation or habitat cover (e.g. 10, or 30, or 70%) that can be applied broadly across different landscapes and different biotic groups – particularly for aggregate measures like species richness (see also Parker and Mac Nally, 2002).

There are several reasons why threshold responses may not be a general phenomenon common to all landscapes, species assemblages, and individual species. First, many studies have shown each species in an assemblage responds differently to landscape change. Some may be area-sensitive, others may be more likely to occur in more (rather than less) sub-divided landscapes. Thus, many species in an assemblage may not respond in the same way to the same landscape variable (e.g. exhibit a sudden change point at 30 % of native vegetation cover). Rather, some might be lost at higher levels, and some at lower levels of native vegetation cover (Radford et al., 2005). A second factor influencing threshold relationships is spatial scale. Landscape change as perceived by humans relates to one spatial scale – which may or may not be important for the particular species or species assemblage of interest. Finally, an absence of threshold effects may occur because some processes like the effect of invasive species can be important in landscapes even well above hypothesized vegetation cover thresholds. For example, the impacts on native vertebrates of feral predators such as the Red Fox (*Vulpes vulpes*) (e.g. Risbey et al., 2000) and other introduced species like the Cane Toad (*Bufo marinus*) (Bennett, 1997) have been pervasive throughout many Australian landscapes including landscapes where vegetation changes have been very limited.

### 3.5. Overview of problems with ecological “short-cuts”

In summary, it is clear there are major deficiencies in each of the “ecological short-cuts” discussed above, which limits their widespread use for gauging the sustainability of forest management (on either public or private forest land). Many other similar approaches (that have not been discussed in this paper) also have substantial problems such as the habitat hectares approach (see McCarthy et al., 2004). These kinds of deficiencies are important because an uncritical acceptance of them may lead a forest manager to believe that s/he is sustainably managing a forest when s/he is not. Given this, together with an existing lack of empirical data on forest biota and the impacts of forestry practices on it, a possible alternative approach is to generate general ecological principles for biodiversity conservation and then embrace these principles through a practical checklist of multi-scaled on-ground management practices. Such general principles and an associated checklist may, in turn, provide a benchmark against which new codes of practice might be developed or the efficacy of existing codes assessed and deficiencies in them subsequently addressed. The following section outlines the general management principles and an allied checklist of management strategies. Notably, such codes and forestry standards typically do not take into account issues outlined in the checklist like the importance of natural disturbance refugia, and the potential problems created by multiple cumulative impacts (e.g. the combination of grazing, logging and fire in forests).
4. What to do given existing limitations? – General principles and a checklist of management strategies

4.1. General principles for managing forest biodiversity

Detailed assessments of the status of biodiversity in almost all parts of the world (e.g. Groombridge, 1992; Andren, 1994; Burgman and Lindenmayer, 1998; State of the Environment, 2001; Groombridge and Jenkins, 2002) indicate that species loss is predominantly driven by habitat loss (reviewed by Primack, 2001; Fahrig, 2003; Lindenmayer and Burgman, 2005). Therefore, the overarching goal of conservation management must be to prevent habitat loss. In forests, conservation planning for many species has often focused on developing strategies that operate at only one or two spatial scales. However, any credible plan for forest biodiversity conservation requires maintaining habitat across the full range of spatial scales. There are five general principles that can help meet this objective:

- The maintenance of connectivity. Connectivity is the linkage of habitats, communities and ecological processes at multiple spatial and temporal scales (Noss, 1991). Connectivity influences key biodiversity conservation processes such as population persistence and recovery after disturbance (e.g. logging, Lamberson et al., 1994), the exchange of individuals and genes in a population (Leung et al., 1993; Saccheri et al., 1998), and the occupancy of habitat patches (Villard and Taylor, 1994; Lindenmayer et al., 2002a).

- The maintenance of landscape heterogeneity. Landscape heterogeneity is a feature of natural forests worldwide; it is the mosaic of patches representing different forest composition and age classes within which different structural conditions occur (Forman, 1995). It is a key general principle for forest biodiversity conservation because the diversity, size, and spatial arrangement of habitat patches is important for many forest taxa (e.g. Hanksi, 1994; Saab, 1999; Debinski et al., 2001).

- The maintenance of stand structural complexity. Structural complexity per se is a common feature of all natural forests throughout the world (Franklin et al., 1981, 2002; Berg et al., 1994; Noel et al., 1998). Stand structural complexity embodies not only particular types of stand attributes, but also the way they are spatially arranged within stands. Such attributes include:
  - Trees from multiple age cohorts within a stand
  - Large living trees and snags
  - Large diameter logs on the forest floor
  - Vertical heterogeneity created by multiple or continuous canopy layers
  - Canopy gaps and anti-gaps (Franklin et al., 2002).

The maintenance of stand structural complexity is critical for forest biodiversity conservation because: (a) It may allow organisms to persist in logged areas from which they would otherwise be eliminated, (b) It can allow logged and regenerated stands to more quickly return to suitable habitat for species that have been displaced, (c) It may enhance dispersal of some animals through a cutover area – a ‘connectivity’ function, and (d) It can provide the within-stand variation in habitat conditions required by some taxa – a ‘habitat heterogeneity’ function.
• The maintenance of the integrity of aquatic systems by sustaining hydrological and geomorphological processes. Aquatic features of forest landscapes — streams, rivers, wetlands, lakes and ponds — are critically important to biodiversity and ecosystem function. The maintenance of the integrity of aquatic ecosystems is a critical component of conservation management because a very large proportion of the biodiversity found in forested landscapes is associated with aquatic ecosystems — including many terrestrial as well as all aquatic organisms (Loyn et al., 1980; Calhoun, 1999; Mac Nally and Soderquist 2002).

• The use of knowledge of natural disturbance regimes in natural forests to guide off-reserve forest management practices. The general philosophy of this principle is that strategies for biodiversity conservation are most likely to be successful in cases where human disturbance regimes (such as logging) are similar in their effects to natural disturbance (Hunter, 1994). Hence, organisms will be best adapted to the disturbance regimes under which they have evolved (Bergeron et al., 1999; Hobson and Schieck, 1999), but potentially susceptible to novel forms of disturbance (or combinations of disturbances) such as those that are more or less frequent and/or more or less intensive than would normally occur (Lindenmayer and McCarthy, 2002). An important aspect of the use of knowledge of natural disturbance regimes is to determine appropriate baselines and ranges of variability against which human disturbance regimes can be compared (Hunter, 1993; Lindenmayer and Franklin, 2002).

Although the maintenance of habitat is the overarching goal of forest biodiversity conservation, habitat suitability is a species-specific construct — what constitutes suitable habitat varies between each species (Morrison et al., 1992; Block and Brennan, 1993; Guissan and Zimmerman, 2000). Similarly, what constitutes suitable connectivity, stand complexity, landscape heterogeneity and aquatic ecosystem integrity will be defined on a species-specific basis and can vary markedly between species (Lindenmayer and Burgman, 2005). Since defining these variables for a large set of species is essentially impossible, enabling or creating spatial and temporal variation in a range of conditions is a practical response to this problem. Therefore, embracing the general principles outlined above will require adopting multiple approaches at multiple scales — to provide conditions needed by different species in at least some parts of a forest landscape (Table 1). Hence, as noted by (Evans and Hibberd, 1990), management for diversity calls for diversity of management. These arguments are the motivation for the checklist presented below. They also highlight the need for a risk-spreading approach to forest management; that is, the importance of not doing the “same thing everywhere” (Lindenmayer and Franklin, 2002; Bunnell et al., 2003).

4.2. A checklist for forest biodiversity conservation

The conservation of forest biodiversity embodies a continuum of conservation approaches from the establishment of large ecological reserves through to an array of off-reserve conservation measures including the maintenance of individual forest structures at the smallest spatial scale (Figure 1). This is demonstrated in the checklist given in Table 2 in which its elements are arranged in approximate hierarchical order progressing from very large-spatial-scale strategies (large ecological reserves such as those spanning a 1000 ha or more) through mid-spatial-scale or landscape-level strategies (e.g. protected areas within production forests) to stand-level silvicultural approaches (e.g. stand structural retention).
<table>
<thead>
<tr>
<th>Principle</th>
<th>Strategy</th>
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<tbody>
<tr>
<td><strong>Principle 1 - Maintenance of connectivity</strong></td>
<td>□  Riparian and other corridors</td>
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<tr>
<td></td>
<td>□  Protection of sensitive habitats with the matrix</td>
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<tr>
<td></td>
<td>□  Vegetation retention on logged areas</td>
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<td></td>
<td>□  Careful planning of roading infrastructure</td>
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<td></td>
<td>□  Landscape reconstruction</td>
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<td><strong>Principle 2 - Maintenance of landscape heterogeneity</strong></td>
<td>□  Riparian and other corridors</td>
</tr>
<tr>
<td></td>
<td>□  Protection of sensitive habitats within the matrix</td>
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<tr>
<td></td>
<td>□  Mid-spatial-scale protected areas (i.e. those 10-100’s of ha)</td>
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<td></td>
<td>□  Spatial planning of areas to be logged</td>
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<td></td>
<td>□  Increased rotation lengths</td>
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<td></td>
<td>□  Landscape reconstruction</td>
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<td></td>
<td>□  Careful planning of roading infrastructure</td>
</tr>
<tr>
<td></td>
<td>□  Use of natural disturbance regimes as templates</td>
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<tr>
<td><strong>Principle 3 - Maintenance of stand complexity</strong></td>
<td>□  Retention of structures and organisms during regeneration harvest</td>
</tr>
<tr>
<td></td>
<td>□  Habitat creation (e.g. promotion of cavity-tree formation)</td>
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<tr>
<td></td>
<td>□  Stand management practices</td>
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<tr>
<td></td>
<td>□  Increased rotation lengths</td>
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<td></td>
<td>□  Use of natural disturbance regimes as templates</td>
</tr>
<tr>
<td><strong>Principle 4 - Maintenance of intact aquatic ecosystems</strong></td>
<td>□  Riparian corridors</td>
</tr>
<tr>
<td></td>
<td>□  Protection of sensitive aquatic habitats with off-reserve areas</td>
</tr>
<tr>
<td></td>
<td>□  Careful planning and maintenance of roading infrastructure</td>
</tr>
<tr>
<td><strong>Principle 5 – Adoption of natural disturbance regimes as templates to</strong></td>
<td>□  Ensuring that strategies are varied between different stands</td>
</tr>
<tr>
<td>guide human disturbance regimes**</td>
<td>□  and landscapes (‘don’t do the same thing everywhere’)</td>
</tr>
</tbody>
</table>
Figure 1 A framework for biodiversity conservation across protected areas (typically in public ownership) and off-reserve areas (including public and private native forests) (redrawn from Lindenmayer and Franklin, 2002)
Table 2  Checklist of factors for off-reserve conservation management (based on Lindenmayer and Franklin, 2002)

<table>
<thead>
<tr>
<th>LARGE ECOLOGICAL RESERVES</th>
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<tbody>
<tr>
<td>□  CARR principles (comprehensive, adequate, representative, replicated) for large ecological reserves and implications for adjacent Private Native Forest lands</td>
</tr>
<tr>
<td>□  Maps of vegetation types for cross-tenure assessment of land uses</td>
</tr>
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<tr>
<th>LANDSCAPE-LEVEL CONSERVATION STRATEGIES WITHIN OFF-RESERVE FOREST</th>
</tr>
</thead>
<tbody>
<tr>
<td>Protected habitat <strong>within the landscape</strong> – protected areas at mid-spatial scales</td>
</tr>
</tbody>
</table>

- □  Special habitats  
  - Cliffs, caves, rockslides etc  
- □  Remnant patches of late-successional forest  
- □  Biological hotspots  
  - Source areas for coarse woody debris, populations of rare species  
- □  Fire, wind and other disturbance refugia  
- □  Aquatic ecosystems and riparian buffers  
  - Springs, seeps, lakes, ponds, wetlands, streams and rivers and associated buffers  
- □  Wildlife corridors  

**Other landscape-level considerations**

- □  Transportation systems (e.g. roading networks)  
- □  Landscape-level goals for specific structural features (e.g. large trees with hollows)  
- □  Spatial and temporal patterns of timber harvesting  
  - Dispersed versus aggregated  
  - Size of harvest units  
  - Rotation lengths  
- □  Restoration and re-creation of late-successional (old growth) forests or other habitat features  
- □  Appropriate fire management regimes (e.g. maintenance of range of post-fire age classes), and varied prescriptions between stands  
- □  Management strategies for particular species (e.g. Swift Parrot)  
- □  Control strategies for unwanted species (e.g. weed management, feral animal control)  
- □  Consideration of natural disturbance regimes as template for logging regimes (e.g. identification of natural disturbance refugia as places for logging exemption)
| ■ Retention of structures and organisms at time of regeneration harvest |
| - Trees with hollows (and recruits), large decaying logs, understorey thickets, gaps and anti-gaps |
| ■ Creation of structural complexity through stand management activities |
| ■ Lengthened rotation times |
| ■ Application of novel silvicultural systems to meet stand-level goals |
| - Variable retention harvest system (VRHS), novel thinning systems |
| ■ Appropriate fire management regimes and varied prescriptions between stands |
| ■ Consideration of adjacency to other vegetation/stands (= landscape context) |
| ■ Management of additional kinds of disturbances (e.g. grazing) |
| ■ Targeted management strategies for particular species |
| ■ Control strategies for unwanted species (e.g. weed management, feral animal control) |
| ■ Consideration of natural disturbance regimes as template for logging regimes |
| - Stand level patterns and quantities of biological legacies that remain after natural disturbance events |
### 4.2.1. Large ecological reserves

The highest level in the hierarchy presented in Figure 1 and Table 1 corresponds to large ecological reserves. They are an essential part of all comprehensive biodiversity conservation plans and are needed in all ecosystems and vegetation types. Large ecological reserves are critically important for at least five key reasons (after Lindenmayer and Franklin, 2002):

- they support some of the best examples of ecosystems, landscapes, stands, habitat, and biota and their inter-relationships as well as opportunities for natural evolutionary processes
- many species find optimum conditions within large ecological reserves which become strongholds for these species
- some species are intolerant of human intrusions, making it imperative to retain some areas which are exempt from human activity
- they provide “control areas” against which the impacts of human activities in managed forests can be compared
- the effects of human disturbance on biodiversity are poorly known and some impacts may be irreversible. Others such as synergistic and cumulative effects can be extremely difficult to quantify or predict. These factors make large ecological reserves a valuable ‘safety net’ relatively free from human disturbance.

Although reserves are an essential component of all conservation plans, it is almost always impossible to create reserve systems that are comprehensive, representative and adequate for all elements of biodiversity (Margules and Pressey, 2000; Scott et al., 2001). This reality is the major limitation of a reserve-only focus and it means that credible plans for forest biodiversity conservation must incorporate off-reserve approaches (including those on private land) that complement reserve-based approaches – i.e. conservation strategies at the landscape and stand levels – as outlined in the sections below (Lindenmayer and Franklin, 2002). Thus, while it is clear and obvious that private native forests would not be part of a formal (public) forest reserve system, they nevertheless have an important conservation role and will need to contribute to the conservation of biodiversity as part of ecologically sustainable forest management.

Mapping of forest types across all tenures and assessing representativeness of forest types in the formal (public) protected area system is fundamental to determining the extent to which off-reserve conservation strategies are needed and the kinds of conservation management activities that will be required. For example, forest types that are poorly protected or not represented in a reserve system will need to be managed differently to forest types that already feature prominently in reserves.

### 4.2.2. Off-reserve conservation measures at the landscape-level

The checklist in Table 1 contains five broad categories of approaches to landscape-level off-reserve forest management. These are:

- establishment of landscape-level goals for retention, maintenance, or restoration of particular habitats or structures (Lindenmayer and Possingham, 1995; Gibbons and Lindenmayer, 2002) as well as limits or thresholds for specific problematic conditions (e.g. the amount of a forest landscape subject to prescribed burning; Gill, 1999)
- the design and subsequent management of transportation systems (generally a road network) to take account of impacts on species, critical habitats, and ecological processes (Forman et al., 2004)
• the selection of the spatial and temporal pattern for harvest units or other management units (Franklin and Forman, 1987)

• the protection of aquatic ecosystems and networks (such as rivers, streams, lakes and ponds), specialized habitats (e.g. cliffs and caves), wildlife corridors, biological hotspots (e.g. spawning habitats, roosting areas for birds or camps for flying foxes), and remnants of late-successional or old-growth forest and disturbance refugia found within off-reserve forests (McCarthy and Lindenmayer, 1999)

• the application of appropriate fire management regimes (Gill, 1999; Keith et al., 2002).

It is important to distinguish between large ecological reserves and the protection of smaller areas within landscapes broadly designated for wood production (the last dot point above). Such mid-spatial scale reserve systems that are managed primarily for conservation are important in several ways including:

• Increasing protection of habitats, vegetation types, and organisms poorly represented or absent in large ecological reserves

• Protecting aquatic and semi-aquatic ecosystems

• Providing refugia for forest organisms that subsequently provide propagules and offspring for recolonizing surrounding forest areas as they recover from timber harvest

• Acting as ‘stepping stones’ to facilitate the movement of biota across managed landscapes.

Notably, although mid-spatial-scale protected areas within wood production areas are a critical component of any forest biodiversity conservation plan, they will not conserve all species – some taxa will require large ecological reserves (Noss and Cooperrider, 1994; Lindenmayer et al., 1999).

Fire management is an additional key aspect of sustainable forest management and biodiversity conservation that is critical at the landscape level (and also the stand-level – see below). Management may involve both the suppression of unwanted (wild) fires and ignition of prescribed fires (Gill, 1999). Objectives will vary depending on the proximity of people and property and the relative importance of values like timber resources, conservation and water production. This is complicated by the fact that few (if any) areas have just one economic or ecological value (Keith et al., 2002). The most appropriate fire regime will depend on the objectives of management and the characteristics of the system, but differences between vegetation communities and individual elements of the biota in their response to fire means that there are no simple management recipes (Whelan, 1995; Lindenmayer and Burgman, 2005). Given such complexity, one management approach is to vary fire regimes between and within landscapes, creating a range of conditions. Therefore, if unsuitable habitats are created in one area, there will be other places where a species can survive. This is consistent with the risk-spreading approach outlined elsewhere in this report (see also den Boer, 1981; see also Lindenmayer and Franklin, 2002).

Perhaps the best informed perspective on landscape-scale fire management comes from Gill (1999, p. 47) who argued that the management of fire for biodiversity conservation should:

“aim at achieving suitable proportions of landscape with a variety of times-since-fire stages within appropriate intensity levels at appropriate times of the year and within appropriate frequency range”.

This is an important objective, but it is not easy to achieve because using fire as a management tool is influenced by many practical constraints, including (in many cases) a limited time window during which prescribed fires can be applied safely (Whelan, 1995), and limited financial and human resources for fire management.
Finally, landscape-level management will be difficult, if not impossible, for the vast majority of private native forest managers to implement. This is because key issues associated with forest landscape management will be influenced by factors outside the boundaries of an individual landholder’s tenure. A coordinating body such as a state government natural resource management agency may be an appropriate agency to provide relevant advice and extension assistance on this issue. Expertise from private organizations is another possible approach.

4.2.3. Off-reserve conservation measures at the stand-level

The finest-scale conservation strategies in the checklist in Table 2 correspond to stand-level off-reserve forest management practices. The objective of off-reserve management at the stand level is to purposefully increase the contribution of harvest units to the conservation of biodiversity. Harvest units can be managed to sustain species, increase habitat diversity, improve connectivity, buffer sensitive areas and sustain ecosystem processes including site productivity.

Developing and maintaining structural complexity in harvested stands is central to any forest management program that has the serious intent of maintaining forest biodiversity and ecosystem processes. This is because the internal structure and composition of harvest units can have a significant influence on the degree to which a managed forest can sustain biodiversity and maintain ecosystem processes. On this basis, four broad types of strategies can contribute to the maintenance of structural complexity:

- structural retention at the time of regeneration harvest (e.g. large hollow trees and associated recruit trees [Gibbons and Lindenmayer, 2002]; understorey thickets [Mueck et al., 1996; Ough and Murphy, 1998], and large fallen logs [Lindenmayer et al., 2002d])
- management of regenerated and existing stands to create specific structural conditions (e.g. through novel kinds of thinning activities [Carey et al., 1999])
- long rotations or cutting cycles (Hunter and Seymour, 1999)
- application of appropriate fire management regimes. Prescribed burning is designed to reduce fuel loads and, in turn, reduce the risk of a high-intensity fire. In some ecosystems, frequent prescribed fires will be necessary to reduce fuel. Fuel is also habitat for animals and plants (e.g. logs that provide nursery sites for plant germination; Howard, 1973; McKenny and Kirkpatrick; 1999). Changes to fuel loads such as logs and leaf litter can therefore impact the elements of the biota that depend on these features (York, 1999; Lindenmayer et al., 2002d). Mosaic or patchy burning can maintain suitable habitat for some ground-dwelling organisms and provide escape routes for others. The interval between burning should be sufficient to regenerate native vegetation and allow fire-sensitive species to recover. However, given limited available data on the impacts of fire on forest ecosystems and associated forest biota (Whelan, 1995), the need for spatial and temporal variability in fire regimes is critical – the risk-spreading approach articulated above.

While each of the various stand-level strategies outlined in this section can make a unique contribution to maintaining biodiversity within managed stands, they also can often be effectively combined to address a broader range of objectives (Taulman et al., 1998). For example, the advantages of long rotations are multiplied when accompanied by structural retention at the time of harvest (Lindenmayer and Franklin, 2002). Conversely, rotation times may be shortened if greater levels of retention characterise logged stands at the time of harvest. Similarly, structural retention without active management of dense regenerating stands can lead to the development of forest conditions with limited habitat value for biodiversity (Franklin et al., 1997; Carey et al., 1999). Innovative silvicultural systems that address the twin objectives of commodity production and biodiversity conservation can be achieved by integrating the above three strategies (e.g. Hickey et al., 2001; Beese et al., 2003; Bunnell et al., 2003; Lindenmayer et al., unpublished data).
4.3 Operationalising the checklist

Clearly in the context of an individual landholder with a private native forest, it would be impossible (and inappropriate) to implement all the elements outlined in the checklist in Table 2 and Figure 1. It is typically not necessary to manage every hectare of a private forest holding for biodiversity values (Salt et al., 2004). Thus, how the components listed in Tables 1 and 2 and Figure 1 are actually addressed in real world forest management — individually and collectively — depends upon many considerations. These include the nature of the landscape (physical and biological conditions), human developments (such as roads), the objectives of the landowner(s), regulatory and social directives, and the species targeted for conservation. Obviously since each landscape is unique in the mix of such considerations, it is impossible to provide generic solutions to landscape- and stand-level prescriptions. Hence, it is impossible to provide specific prescriptions for the on-ground implementation of the items in the checklist. Therefore, the best and/or most appropriate strategies will vary according to particular conservation and other management objectives within particular areas. For example, the kinds of silvicultural options and strategies relevant to the wet eucalypt forests of Tasmania and its associated biota will be quite different even for broadly similar forest types on the mainland (e.g. in the Central Highlands of Victoria) in part because of the significant differences in biodiversity (such as hollow-dependent vertebrates) between the two regions. Similarly, what constitutes suitable habitat or connectivity for a given species in a particular landscape dominated by a particular forest type or set of forest types can vary markedly in another landscape even for the same species (e.g. Greater Glider [Petauroides volans] Lindenmayer, 2002a, 2002b). Many other examples of such differences and the problems they create could be cited. The implications are that it is not feasible to recommend a series of simple measures that can be routinely made that are relevant across all forest types and forest landscapes within which private native forest harvesting may occur. It is essential then, for regional forest managers (such as those from government agencies and catchment management authorities) to be quite clear about the vision they have for a forest landscape or a landscape mosaic that includes areas of forest, and also to have a vision about the stands which comprise that landscape. Explicit statements of these objectives then need to be complemented by the development of strategies and practical tools to achieve the stated objectives.

The simple checklist presented in this report offers a hierarchical framework of general strategies for consideration in meeting these objectives. It is notable that current codes of forestry practice and forestry standards typically do not take into account some of the issues and management needs/considerations that have been outlined in the checklist. Four key examples are:

- the importance of using natural disturbance regimes as templates to guide human (logging) disturbance regimes to better ensure that forest ecosystems are managed within the natural bounds of disturbance intensity and variability (Lindenmayer and McCarthy, 2002). Biodiversity conservation and other measures of forest sustainability are more likely to be maintained when human disturbance regimes do not fall outside the bounds of natural disturbance regimes (Attiwill, 1994; Lindenmayer and Franklin, 2002)

- the importance of the maintenance of natural disturbance refugia for biodiversity conservation (e.g. multi-aged stands in Victorian ash forests; see Mackey et al., 2002), and the careful management of these areas to ensure their integrity is not impaired as an outcome of additional human disturbances

- the need for careful management of areas subject to forest management to limit potential problems created by multiple cumulative impacts (e.g. the combination of grazing, logging and fire) on biodiversity and stand structural complexity; see Smith et al., 2002; McAlpine et al., 2002a)

- the need to better develop post-fire salvage logging prescriptions to ensure that the potentially negative impacts of wildfire events on biodiversity and forest structure and integrity are not magnified by additional human disturbance (Lindenmayer et al., 2004; Lindenmayer and Ough, 2005).
One good way to start to better integrate production and conservation on a private native forest is to develop stand-level plans and maps and landscape plans and maps. This recognises the reality that not all parts of a private forest holding can be managed to maximize conservation objectives. Rather, some places will be more important and others less important – for example, it may prove to be best to concentrate habitat conservation around riparian areas which typically need to be set aside for water quality reasons and that also support higher levels of species richness than elsewhere in a landscape (Loy et al., 1980; Mac Nally and Soderquist, 2000). However, some elements of the biota do not occur in riparian zones and other parts of forest landscapes will also need to be appropriately managed to maintain populations of such taxa (Claridge and Lindenmayer, 1994). In other cases, levels of retained trees on logged sites might be higher in particular areas (such as those that also support a well developed understorey) than elsewhere in a landscape because of the superior habitat quality, for many species, of areas with a combination of forest understorey vegetation and large hollow-bearing trees (Gibbons and Lindenmayer, 2002). Therefore, whole-property plans that encompass many different stands across a range of landscapes can help set priorities for deciding which actions would be best applied on what parts of a land holding (Salt et al., 2004). Many examples are available from the literature on traditional farm management (e.g. Campbell, 1991; Goldney and Wakefield, 1997) and these can be readily applied to private native forests in the context of assigning different kinds of management strategies on different parts of a forest holding. However, issues remain concerning links between management at the stand and multi-stand level on an individual private native forest holding and how such management practices relate to larger landscape scales spanning different properties and areas with different tenures (see Section 5). Conversely, in many cases the response of some elements of the biota at a stand level will be strongly influenced by ecological processes and spatial patterns operating at larger spatial scales (such as at the landscape scale) (Lindenmayer and Fischer 2006). Spatial relationships between the stand and landscape levels can be complex and appropriate management regimes will often require careful co-ordination by regional or other bodies. Expert advice from, and co-ordination roles played by, State and Territory Government agencies (including catchment management authorities) may be important in this regard.
5. Other key issues

5.1. The need for multiple, multi-scaled conservation measures

The definition of biodiversity given above highlights it as a multi-scaled concept. In parallel with this, attempts to conserve forest biodiversity must also be multi-scaled – with appropriate conservation strategies involving considerations ranging from those at the individual tree-level through to landscape and regional levels. This is implicit in the checklist which highlights the links between multiple management scales and the multi-scaled nature of biodiversity. Multi-scaled management is needed because:

- there are multiple ecological scales for different ecological processes (Urban et al., 1987; Poff, 1997; Elkie and Rempel, 2001). For example, the ecological process of habitat loss can occur at regional and landscape levels by activities such as forest clearing (Angelstam, 1996). Particular age classes (e.g. old growth) can be subject to habitat loss and fragmentation within areas of formerly continuous forest cover. Finally, structural and floristic attributes can be lost from individual stands (Angelstam, 1996).

- there are multiple ecological scales for different species (Allen and Hoekstra, 1992). For example, the spatial requirements of invertebrates requiring decayed logs with particular sorts of attributes (e.g. Velvet Worms [Phylum Onycophora] Barclay et al., 1999) are markedly different from the spatial requirements of wide-ranging predators such as large forest owls (Lamberson et al., 1994).

- there are multiple ecological and management scales for the same species (Forman, 1964; Hokit et al., 1999). Lindenmayer (2000) demonstrated this for Leadbeater’s Possum (Gymnobelideus leadbeateri) in the Central Highlands of Victoria – key management actions were required at the individual tree level, the stand level, the patch level, the landscape level and the regional level. This outcome for Leadbeater’s Possum is paralleled by many other examples for forest landscapes around the world ranging from tropical forests in New Guinea (Diamond, 1973) to the temperate forests in the Bavarian Alps of Germany (Storch, 1997).

Thus, there is no single ‘right’ or ‘sufficient’ scale for forest and conservation management. A single conservation strategy adopted at a single spatial scale will provide suitable habitat for only a limited number of different taxa and may be insufficient for some species. If, for example, off-reserve management is practiced only at the stand level, key landscape level values may be lost. Indeed, many processes like the movement of birds and bats and the migration of fish within forest streams and rivers cannot be dealt with at the stand level. Similarly, a sole focus at a landscape scale may fail to provide the appropriate combinations of habitat elements needed by many species within individual stands (Morrison et al., 1992; Lindenmayer and Franklin, 1997). Therefore, the adoption of multiple strategies at multiple spatial scales is important because it increases the chances of satisfying the general principles for forest conservation touched on above. That is, suitable connectivity, heterogeneity, stand complexity and aquatic ecosystem integrity (see Table 1) will be provided for most taxa in at least some parts of a forest estate.

Although it is clear there are multiple management scales, the stand level scale is the one that will be most relevant to managers of private native forests. Expert assistance in terms of the kinds of silvicultural options available to meet particular stand-level objectives (including the marriage of production and conservation ones) may need to be provided to private native forest managers to meet such goals. As outlined above in the section on landscape-level goals and objectives, it remains unclear how such expert advice might be tended, although extension work through state government bodies or via relevant private organisations is one possible approach. Similarly, it remains unclear how management at the stand level within private native forests can be best integrated with the need for landscape-level management which, by definition, will require consideration of issues across different
properties and across areas with different tenures. However, again it may be possible that State and Territory Government agencies might play an important co-ordinating and expert advisory role in this regard.

5.2 The need for multiple, multi-scaled conservation measures – risk-spreading

The implementation of an array of differently-scaled strategies is a risk-spreading approach. That is, if one strategy subsequently proves to be ineffective others will be in place that might better conserve the entities targeted for management. Hence, risk-spreading reduces the over-reliance on any one particular conservation strategy and attempts to deal with the considerable uncertainty and lack of knowledge of the effectiveness of current conservation management strategies. Risk-spreading is particularly appropriate for biodiversity conservation because it is often extremely difficult to accurately forecast the response of species to processes such as landscape modification (see Mac Nally et al., 2000), stand simplification (Lindenmayer et al., 2002d) or prescribed fire (Whelan, 1995; Whelan et al., 2002). For example, in the case of fire, a fire regime which benefits a particular species or set of species may not suit others. Indeed, as Keith et al. (2002) noted:

“the full spectrum of biodiversity cannot be maintained if fire management only addresses a single species in isolation”.

5.3. The need for multiple, multi-scaled conservation measures – cumulative negative impacts vs. synergies

Ignoring the need for an array of different strategies can lead to compounding or cumulative negative impacts for biodiversity. For example, the loss of structural complexity within stands can accumulate over many cutover sites and result in homogenized landscapes (Gibbons and Lindenmayer, 2002). Conversely, an advantage of multiple management strategies is that a given approach may generate positive benefits for another strategy implemented at a different spatial scale. For example, smaller cutover units can reduce rates of windthrow and vegetation loss in adjacent wildlife corridors, and riparian areas within the production forests (Lindenmayer et al., 1997). Other work demonstrates positive synergistic benefits for biodiversity from implementing several strategies concurrently such as the implementation of stand retention over multiple lengthened rotations (Franklin et al., 1997).

The above examples indicate that different multi-scaled approaches do not act in isolation. Different combinations of management strategies or differences in the relative emphasis on particular strategies can sometimes achieve the same objectives for biodiversity conservation. This will sometimes (although not always) allow for trade-offs between such strategies. Some hypothetical examples are:

- increased levels of stand retention such as clumps of retained trees and understorey thickets at the time of regeneration harvest may mean that the size of cutover units can be increased or the levels of stand management reduced
- increased levels of stand retention may make it possible to reduce rotation times
- lengthened rotation times with stand retention may allow requirements for strictly protected wide wildlife corridors to be relaxed.

5.4. The need for true adaptive management and dedicated rigorous monitoring

Although the general ecological principles and associated checklist are intuitive, there are limited data to determine the effectiveness of many particular on-the-ground management actions. Indeed, the more
than 75 major enquiries into the Australian timber industry since World War II have highlighted the need for basic information on Australian forests, including data on the impacts of forest management on biota (e.g. Resource Assessment Commission, 1992). Thus, limited data and high levels of uncertainty are characteristic of all managed forests, including those on private land. Considerable effort is needed to adopt truly adaptive management “natural experiments” and monitoring to: (1) better identify the impacts of logging operations and other kinds of management on biodiversity, and (2) quantify the efficacy of impact mitigation strategies and ways to improve them where necessary. Unfortunately, the record on forest monitoring (and particularly on biodiversity) remains extremely poor both on public native forests and on private forested land in all Australian jurisdictions. A commitment to true adaptive management, that involves rigorous monitoring and a commitment to management change when negative impacts are identified, should be considered to be a key element of ecologically sustainable forest management (Lindenmayer et al., 2000).

There are several key issues associated with rigorous monitoring and adaptive management. First, in many cases, the expertise to conduct monitoring will lie with government organisations such as state-based forestry and natural resource management agencies and it may be appropriate for them to engage in such monitoring roles. Second, monitoring is a time and data-intensive process that can be expensive to conduct. Therefore, careful consideration needs to be given to ensure the magnitude of monitoring responsibilities is consistent with the size and intensity of associated forestry activities (Franklin et al., 1999) including those on private native land. Third, it is neither possible nor desirable to monitor everything. Rather, it is better to monitor a few entities well than many things poorly and hence monitoring programs need to be well focused and carefully planned and executed (Lindenmayer, 1999).
6. Summary and conclusions

A major overarching theme in forest biodiversity conservation must be to maintain habitat suitability for forest biota. This is because habitat loss is a key factor underpinning both the loss of biodiversity *per se* and the loss of species in forest ecosystems. Several general principles can help meet the critical goal of maintaining habitat across the full range of spatial scales. These are:

- the maintenance of connectivity
- the maintenance of landscape heterogeneity
- the maintenance of stand structural complexity
- the maintenance of the integrity of aquatic ecosystems by sustaining hydrological and geomorphological processes
- the use of knowledge of natural disturbance regimes in natural forests to guide off-reserve forest management practices.
- the adoption of a risk-spreading strategy in which forest managers do not “do the same thing everywhere”.

Developing management plans and strategies consistent with the five general principles listed above and the general overarching principle of habitat maintenance is a complex, challenging, and multi-scaled task for forest managers. In attempting to meet this challenge, there are three broad groups of conservation strategies that together form part of any credible approach to forest biodiversity conservation. These are:

1. setting aside large ecological reserves
2. adopting landscape-level off-reserve management strategies
3. adopting stand-level off-reserve management strategies.

Based on these three broad categories of strategies, a simple, hierarchically-arranged checklist of conservation strategies is presented to guide the development of forest conservation plans, including those used in private native forests. The checklist is intuitive but its elements appear not to have previously been presented. However, the elements of the checklist are likely to be useful as a benchmark against which current or planned codes of practice can be assessed. Strategies at the stand-level, in particular, relate most strongly to individual landholders with areas of private native forest. However, stand-level management must also be carefully considered in relation to landscape-level factors – a problem that will require cross-property and cross-tenure co-ordination. Organizations such as State Government agencies and catchment management authorities are those most likely to be important in this regard.
7. References


Private native forests comprise a significant proportion of the cover of native forest in many Australian jurisdictions. As such, their sustainable management is a critical issue for the maintenance of forest productivity and forest biodiversity.

In this report, a series of ecological principles is outlined to guide how forestry on private land might best achieve biodiversity conservation. A checklist of on-ground management strategies aimed at operationalising these principles is described. A discussion of the meaning of ecologically sustainable forest management, together with an outline of problems associated with biodiversity conservation “short-cuts” is given to provide suitable context to the description of the principles and associated checklist.

Importantly, the focus of this paper is on existing areas of native forest rather than exotic plantations (including eucalypt plantations) or farm forests that have been established on former grazing lands.

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