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General management principles and a checklist of strategies to guide forest biodiversity conservation

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ABSTRACT

Many indicators and criteria have been proposed to assess the sustainable management of forests but their scientific validity remains uncertain. Because the effects of forest disturbances (such as logging) are often specific to particular species, sites, landscapes, regions and forest types, management "shortcuts" such as indicator species, focal species and threshold levels of vegetation cover may be of limited generic value. We propose an alternative approach based on a set of five guiding principles for biodiversity conservation that are broadly applicable to any forested area: (1) the maintenance of connectivity; (2) the maintenance of landscape heterogeneity; (3) the maintenance of stand structural complexity; and (4) the maintenance of aquatic ecosystem integrity; (5) the use of natural disturbance regimes to guide human disturbance regimes.

We present a checklist of measures for forest biodiversity conservation that reflects the multi-scaled nature of conservation approaches on forested land. At the regional scale, management should ensure the establishment of large ecological reserves. At the landscape scale, off-reserve conservation measures should include: (1) protected areas within production forests; (2) buffers for aquatic ecosystems; (3) appropriately designed and located road networks; (4) the careful spatial and temporal arrangement of harvest units; and (5) appropriate fire management practices. At the stand level, off-reserve conservation measures should include: (1) the retention of key elements of stand structural complexity (e.g., large living and dead trees with hollows, understorey thickets, and large fallen logs); (2) long rotation times (coupled with structural retention at harvest); (3) silvicultural systems alternative to traditional high impact ones (e.g., clearcutting in some forest types); and (4) appropriate fire management practices and practices for the management of other kinds of disturbances.

Although the general ecological principles and associated checklist are intuitive, data to evaluate the effectiveness of many specific on-the-ground management actions are limited. Considerable 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 activities on biodiversity, and; (2) quantify the effectiveness of impact mitigation strategies; and (3) identify ways to improve management practices.

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1. Introduction

Forests support about 65% of the world's terrestrial taxa (World Commission on Forests and Sustainable Development, 1999) and have the highest species diversity for many taxonomic groups including birds (Gill, 1995), invertebrates (Erwin, 1982; Majer et al., 1994) and microbes (Torsvik et al., 1990; Crozier et al., 1999). Conserving forest biodiversity is therefore a critical task (Aanderaa et al., 1996; Hunter, 1999; Putz et al., 2000) and has rightly become a key component of many national and international forest management agreements (e.g. Commonwealth of Australia, 1998, 2001; Montréal Process Liaison Office, 2000; Food and Agriculture Organisation of the United Nations, 2001).

Most programs to sustain forest biodiversity have focused on the creation of protected areas. Reserves are a critical part of any credible strategy for conserving forest biodiversity (Norton, 1999), but reserves alone are insufficient to adequately conserve forest biodiversity (Sugal, 1997; Daily et al., 2001; Lindenmayer and Franklin, 2002), in part because 92% of the world's forests are outside formally protected areas (Commonwealth of Australia, 1999).

In this paper, we outline a series of ecological principles to guide how forestry practices might best achieve biodiversity conservation. A checklist of on-ground management strategies aimed at operationalising these principles is described. Problems associated with biodiversity conservation "short-cuts" are outlined to provide a context for the description of the ecological principles and an associated checklist of strategies. We focus on existing areas of native forest rather than exotic plantations or farm forests that have been established on former grazing lands. Biodiversity conservation in plantations and farm forests is considered elsewhere (e.g. Peterken and Ratcliffe, 1995; Moore and Allen, 1999; Lindenmayer and Hobbs, 2004; Salt et al., 2004). In addition, much of our focus is on "well-forested" landscapes such as those in North America, South America and Australia where "western-style" exploitation has been imposed only relatively recently. However, many of the ecological principles and elements in the checklist should be relevant to forest management in other places such as Europe (e.g. Peterken, 1996; Angelstam, 1996; Fries et al., 1997). Finally, our discussion is restricted to sustaining populations of native forest biota, although many other factors (such as maintaining soil fertility and productivity) are also fundamental aspects of ecologically sustainable forest management.

2. Background – ecologically sustainable forest management and biodiversity conservation

We define ecologically sustainable forestry as:

> *"...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 within the bounds of normal disturbance regimes."*

Achieving ecological sustainability will often require determining appropriate baselines and ranges of variability for natural disturbance regimes against which human disturbance regimes can be compared (Hunter, 1993; Rülcker et al., 1994). The conservation of biodiversity is clearly part of perpetuating ecosystem integrity (as highlighted in the above definition). There are numerous definitions of biodiversity – Delong (1996) and Bunnell (1998) reviewed approximately 90 interpretations of the concept. For this paper, biodiversity is considered to encompass:

> *...genes, individuals, demes, populations, metapopulations, species, communities, ecosystems, and the interactions between these entities.*

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

Nevertheless, several international and national initiatives have sought to develop criteria and indicators of sustainability in forests (e.g. Arborvitae, 1995) despite the problems of: (1) defining biodiversity; (2) determining what constitutes ecologically sustainable forestry; and (3) the sheer impossibility of measuring and monitoring the impacts on all species of various management practices. Some kinds of indicators of sustainability will be essential, but the scientific validity of most indicators of biodiversity is poor. This is one reason we have proposed the set of guiding principles and the checklist presented later in this paper.

3. Problems with "short-cut" methods that attempt to promote ecologically sustainable forest management practices

"Short-cuts" aimed at promoting biodiversity conservation as part of the sustainable management of natural resources include indicator species, focal species, and thresholds in levels of native vegetation cover. All of these (and others; e.g. see McCarthy et al., 2004) have problems as discussed below.

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 is widely used as a measure of ecologically sustainable forest management. Many taxonomic groups have been suggested as indicators (reviewed by Lindenmayer and Burgman, 2005), while problems with the indicator species concept have been reviewed thoroughly by Landres et al. (1988), Temple and Wiens (1989), Niemi et al. (1997), Simberloff (1998), and Lindenmayer et al. (2000). The chief difficulties include:

- An absence of documented causal relationships between indicator species and the entities they are assumed to indicate.
- Major variation between species in their response to environmental change including members of the same guild or closely related species.
- The insensitivity of indicator species to some types of significant environmental change, and
- Insufficient knowledge to guide the selection of indicator species. For example, although many workers have proposed particular taxa as indicator species (e.g. Davey, 1989; Johnson, 1994; Hill, 1995), the associated organisms or other entities for which they are supposed to be surrogates often are not identified (Lindenmayer and Cunningham, 1997).

Much more work needs to be done to confirm the relationships between species chosen as indicators and environmental change (Simberloff, 1999; Fleishman et al., 2002; Kavanagh et al., 2004). This does not mean that particular species should not continue to be monitored or be the target of management actions (reviewed by Thompson and Angelstam, 1999). However, it should not be assumed that the response of a given species (e.g. to logging or targeted habitat management) will necessarily be a good surrogate for the responses of other taxa.

3.2. Focal species

Focal species are defined as those most influenced by threatening processes, for example, the taxon or taxa most limited by dispersal abilities, resources or ecological processes (Lambeck, 1997). A landscape may then be managed for a suite of focal species, each of which is thought to be sensitive to a particular threatening process. There are serious flaws in the focal species approach (Lindenmayer et al., 2002a). Like other taxon-based surrogate schemes, a suite of focal species is presumed to act collectively as a surrogate for other elements of the biota, but a landscape managed for a given set of focal species may not meet the requirements of the remaining biota. The focal species approach also may be difficult to apply because of the lack of science to guide the selection of a reasonably reliable set of focal species in the majority of landscapes. 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 tackle the difficult process of landscape management (Lindenmayer and Fischer, 2003).

3.3. Thresholds of vegetation cover

With and Crist (1995) 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 (Rolstad and Wegge, 1987; Andrén, 1994, 1999; Enoksson et al., 1995; With and Crist, 1995). The search for threshold responses for biota is an increasing focus of biodiversity-related research (e.g. McAlpine et al., 2002a; Radford and Bennett, 2004) including forest management research. They could be a valuable tool for use in landscape management such as in planning the extent and spatial arrangement of use of natural resources, but it seems possible that threshold responses for aggregate measures such as species richness may not exist in some ecosystems (Parker and Mac Nally, 2002; Lindenmayer et al., 2005). Several reasons may account for this:

- Patterns of vegetation loss do not occur in a random fashion (Saunders et al., 1987) which results in vegetation cover patterns comprised of highly varying levels of habitat quality (e.g. a distinct bias toward low productivity habitats of limited value for human uses). Vegetation cover levels may therefore equate poorly to levels of habitat suitability for many assemblages and elements of the biota.
- Each species in an assemblage responds differently to landscape change. For example, some may be more (rather than less) likely to occur in 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). Notably, some species might be lost at higher levels, and some at lower levels of native vegetation cover (Radford et al., 2005).
- Other factors such as invasive pest and/or weed species may have a large impact on the distribution and abundance of many members of an assemblage even at vegetation levels well above hypothesized vegetation cover thresholds.

Thresholds will exist in some landscapes, but it seems 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 (Parker and Mac Nally, 2002). Major changes may occur across a broad band of points and the threshold concept might be better re-described as a regime shift – a phenomenon for which there is good empirical evidence (Folke et al., 2004).

3.4. Overview of problems with ecological "short-cuts"

All of the "ecological short-cuts" exhibit deficiencies which limit their widespread use in gauging the sustainability of forest management. Uncritical use of such short-cuts therefore may lead forest managers to believe that a forest is being managed sustainably when it is not. Therefore, we propose an alternative approach of general ecological principles for biodiversity conservation and 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. These principles and a checklist are outlined in the following section.

4. General principles for managing forest biodiversity

Species loss is predominantly driven by habitat loss (reviewed by Groombridge and Jenkins, 2002; Primack, 2001; Fahrig, 2003). Therefore, the overarching goal of conservation management must be to prevent habitat loss. Forest biodiversity conservation will depend on 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).
- *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. 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 (Aapala et al., 1996; Calhoun, 1999; Soderquist and Mac Nally, 2000).
- *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 (Franklin and van Pelt, 2004). Attributes contributing to stand structural complexity include: (1) Trees from multiple age cohorts within a stand. (2) Large living trees and snags (Linder and Östlund, 1998). (3) Large diameter logs on the forest floor (Harmon et al., 1986; Berg et al., 1994). (4) Vertical heterogeneity created by multiple or continuous canopy layers (Brokaw and Lent, 1999). (5) Horizontal heterogeneity of which canopy gaps and anti-gaps are examples (Franklin and van Pelt, 2004). The maintenance of stand structural complexity is critical for forest biodiversity conservation because it may allow organisms to persist in logged areas from which they would otherwise be eliminated. It also may facilitate a more rapid return of logged and regenerated stands to suitable habitat for species that have been displaced. The maintenance of stand structural complexity may enhance dispersal of some animals through a cutover area – a 'connectivity' function. Finally, structural complexity can provide the within-stand variation in habitat conditions required by some taxa – a 'habitat heterogeneity' function.
- *The maintenance of landscape heterogeneity*. Ecosystems are naturally heterogeneous and landscape heterogeneity is a feature of natural forests worldwide. Disturbance regimes may create heterogeneous land cover, such as different successional stages in different locations following a wildfire (Whelan et al., 2002). In addition, landscapes are characterized by natural environmental gradients (e.g. in topography, climate, or soil type and soil depth; see Austin and Smith, 1989). Landscape heterogeneity corresponds to the mosaic of patches representing different forest composition and age classes within which different structural conditions occur (Forman, 1995). Different species inhabit different environmental conditions in natural landscapes and the diversity, size, and spatial arrangement of habitat patches is important for many taxa (e.g. Hanski, 1994; Saab, 1999; Debinski et al., 2001).
- *The use of knowledge of natural disturbance regimes in natural forests to guide off-reserve forest management practices*. 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, 1993; Korpilahti and Kuuluvainen, 2002); such as, for example, the kinds and numbers of biological legacies (sensu Franklin et al., 2000) and the spatial patterns of environmental conditions (e.g. patch types) remaining after disturbance (Delong and Kessler, 2000). Organisms are likely to be best adapted to the disturbance regimes under which they have evolved (Bergeron et al., 1999; Hobson and Schieck, 1999), but are 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). Natural disturbance regimes may therefore be appropriate baselines and ranges of variability against which human disturbance regimes can be compared (Hunter, 1993; Angelstam et al., 1995).

4.1. *Relevance to landscapes with a long history of "western-style" human management*

As set out at the start of this paper, the ecological principles listed in the previous section are directed primarily at landscapes with a relatively recent history of "western-style" exploitation. While some of our principles will be relevant to forest landscapes such as those in many parts of Europe with a prolonged history of management and extensive fragmentation (Fries et al., 1997), others may not. For example, the fundamental importance of reserves is recognized in all jurisdictions (Margules and Pressey, 2000). Similarly, the creation and/or maintenance of stand structural complexity is essential for biodiversity conservation in all forests, including those with a long history of management (e.g. Linder and Östlund, 1998). In addition, landscape heterogeneity is preferable to intensive forest management resulting in landscape homogeneity (Lindenmayer and Fischer, in press). While the maintenance of landscape heterogeneity is best guided by an understanding of natural heterogeneity in a given landscape, some landscapes have long since lost their natural disturbance regimes (Zackrisson, 1977) and natural patterns of heterogeneity (Gustaffson et al., 1999). Indeed, many European landscapes exemplify this situation (Peterken, 1996). Notably, landscape heterogeneity tends to benefit native species richness in such situations, even if it is not based on natural heterogeneity patterns that once prevailed (e.g. Ferreras, 2001; Palomares, 2001).

Table 1 – Management strategies to achieve general biodiversity conservation principles (based on Lindenmayer and Franklin, 2002)

Principle	Strategy
Principle 1 – maintenance of connectivity	Riparian and other corridors Protection of sensitive habitats within the matrix Vegetation retention on logged areas throughout the landscape Careful planning of roading infrastructure Landscape reconstruction
Principle 2 – maintenance of landscape heterogeneity	Riparian and other corridors Protection of sensitive habitats within the matrix Mid-spatial-scale protected areas Spatial planning of cutover sites Increased rotation lengths Landscape reconstruction Careful planning of roading infrastructure Use of natural disturbance regimes as templates
Principle 3 – maintenance of stand complexity	Retention of structures and organisms during regeneration harvest Habitat creation (e.g. promotion of cavity-tree formation) Stand management practices Increased rotation lengths Use of natural disturbance regimes as templates
Principle 4 – maintenance of intact aquatic ecosystems	Riparian corridors Protection of sensitive aquatic habitats with off reserve areas Careful planning and maintenance of roading infrastructure
Principle 5 – adoption of natural disturbance regimes as templates to guide human disturbance regimes	Ensuring that strategies are varied between different between stands and landscapes (‘do not do the same thing everywhere’)

4.2. *Spatial and temporal variation in conditions as a risk-spread strategy*

The maintenance of habitat is the overarching goal of forest biodiversity conservation, but what constitutes suitable habitat is different for each species (Morrison et al., 1992; Block and Brennan, 1993; Guisan and Zimmerman, 2000). Similarly, what constitutes suitable connectivity, stand complexity, landscape heterogeneity and aquatic ecosystem integrity will also be defined on a species-specific basis and can vary markedly between species. Enabling or creating spatial and temporal variation in a range of conditions at multiple spatial scales is a practical response to the problem of defining these variables for a large set of species. Conditions needed by different species should then be provided in at least some parts of a forest landscape (Table 1). Management for diversity calls for diversity of management (Evans and Hibberd, 1990) and this a risk-spreading approach to forest management to ensure not doing the “same thing everywhere” (Lindenmayer and Franklin, 2002).

5. 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 (Fig. 1). In Table 2 the elements are arranged in approximate hierarchical order progressing from regional-scale strategies (large ecological reserves) through landscape-level strategies (e.g. protected areas within production forests) and stand-level silvicultural approaches (e.g. stand structural retention).

5.1. *Large ecological reserves*

Large ecological reserves are an essential part of all comprehensive biodiversity conservation plans and 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 only 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 largely exempt from human activity.
- Large ecological reserves 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.

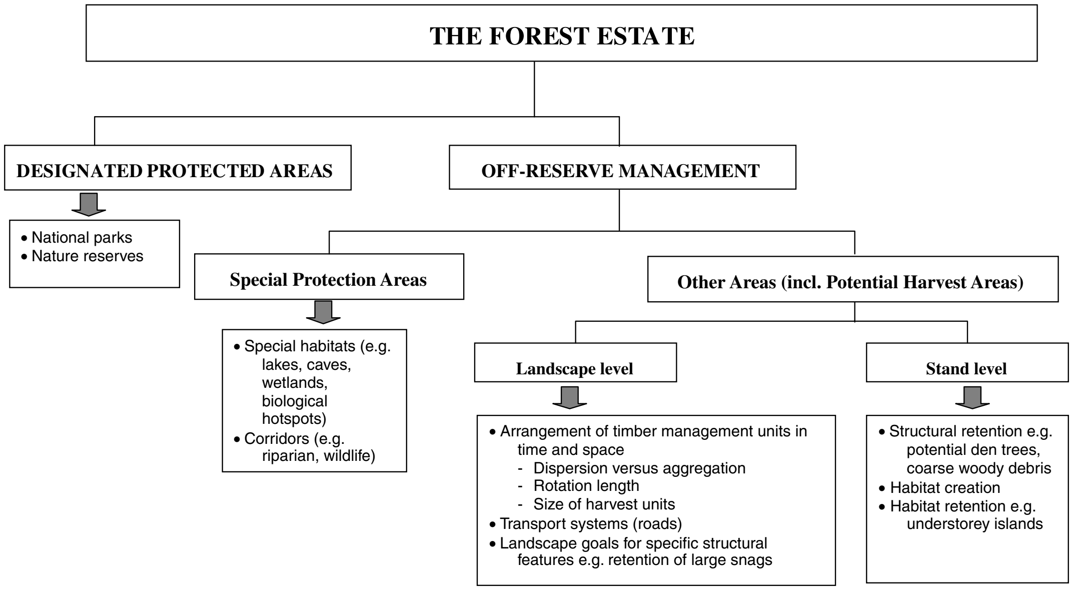

Fig. 1 – A framework for biodiversity conservation across protected areas (typically in public ownership) and off-reserve areas (including public and private native forests) (redrawn from Lindenmayer and Franklin, 2002).

Large ecological reserve systems are rarely comprehensive, representative and adequate for all elements of biodiversity (Margules and Pressey, 2000; Scott et al., 2001). In other cases, past land management means there are few or no opportunities to set aside large ecological reserves (e.g. in parts of southern Sweden; Gustaffson et al., 1999). Hence, credible plans for forest biodiversity conservation must incorporate off-reserve approaches that complement reserve-based approaches – i.e. conservation strategies at the landscape and stand levels (Lindenmayer and Franklin, 2002).

Mapping of forest types across all tenures and assessing representativeness of forest types in the formal (public) protected area system should reveal 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 in a reserve system will need to be managed differently than forest types already well represented in reserves.

5.2. *Off-reserve conservation measures at the landscape-level*

The five broad categories of approaches to landscape-level off-reserve forest management are:

- Establishment of landscape-level goals for retention, maintenance, or restoration of particular habitats or structures as well as limits to 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., 2002).
- The selection of the spatial and temporal pattern for harvest units or other management units (Franklin and Forman, 1987).
- The application and/or management of appropriate disturbance regimes such as those involving fire (Rülcker et al., 1994; Keith et al., 2002) or grazing (Vera, 2000).
- 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).

It is important to distinguish between large ecological reserves (Noss and Cooperrider, 1994) and the protection of smaller areas *within* landscapes broadly designated for wood production (Gustaffson et al., 1999). Such systems of scattered small reserves provide: (1) increased protection of habitats, vegetation types, and organisms poorly represented or absent in large ecological reserves; (2) protection for aquatic and semi-aquatic ecosystems; (3) refugia for forest organisms that subsequently provide propagules and offspring for recolonising surrounding forest areas as they recover from timber harvesting; and, (4) 'stepping stones' to facilitate the movement of biota across managed landscapes.

The management of disturbances such as fire is an additional key aspect of landscape-level sustainable forest management and biodiversity conservation. This may involve both the suppression of unwanted (wild)fires and ignition of prescribed fires (Gill, 1999). Issues associated with the impacts of disturbances by fire and its effects on biota are complex. This is because in some landscapes, such as those in Sweden, problems like a lack of regeneration of particular plant species have been created by the absence of fire (Zackrisson, 1977) whereas in others such as Ponderosa Pine (*Pinus ponder-*

Table 2 – Checklist of factors for off-reserve conservation management (based on Lindenmayer and Franklin, 2002)

Large ecological reserves
CARR principles (comprehensive, adequate, representative, replicated) for large ecological reserves and implications for adjacent Private Native Forest lands
Maps of vegetation types for cross-tenure assessment of land uses
Landscape-level conservation strategies within off-reserve forest
Protected habitat within the landscape – protected areas at intermediate-spatial scales
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 a 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)
Stand-level conservation strategies within off-reserve forest
Habitat within management units or stands
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

osa) stands in south-western USA, fire suppression together with livestock grazing mean that wildfires are too intense (Covington et al., 1997; Moore et al., 2004). The objectives of fire management will vary depending on the proximity of people and property and the relative importance of values like timber resources, conservation and water production (Keith et al., 2002). The most appropriate fire regime also will depend on the characteristics of the system (Rülcker et al., 1994). Differences between vegetation communities and individual elements of the biota in their response to fire mean that there are no simple management recipes (Whelan, 1995). Fire regimes may be varied between and within landscapes, to create a range of conditions. Therefore, if unsuitable habitats are created in one area for a given species, there will be other places it can survive. Gill (1999, p. 47) 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".*

5.3. *Conservation measures at the stand-level*

The objective of off-reserve management at the stand level is to increase the contribution of harvest units to the conservation of biodiversity. Harvest units can be managed to: (1) sustain species; (2) increase habitat diversity; (3) improve connectivity; (4) buffer sensitive areas, and, (5) sustain ecosystem processes including site productivity.

The internal structure and composition of harvested units can have a significant influence on the degree to which a managed forest can sustain biodiversity and maintain ecosystem processes. Several 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 (Fries et al., 1997); understorey thickets (Ough and Murphy, 1998), and large fallen logs (Harmon et al., 1986)). In other cases, specifically targeted strategies may be required to add or create particular structures such as girdling trees to increase quantities of dead wood (Bull and Partridge, 1986) or installing nestboxes (Petty et al., 1994).
- Management of regenerated and existing stands to create specific structural conditions (e.g. through novel kinds of thinning activities (Carey et al., 1999a)). This may include the maintenance of open areas as well as heath and grassland habitats within forests that can be critical for some key elements of biota. For example, in the forests of the Swiss Jura, a reduction in the cover of trees and shrubs is considered critical for the survival of populations of the Asp Viper (*Vipera aspis*) (Jäggi and Baur, 1999).
- Long rotations or cutting cycles (Seymour and Hunter, 1999).
- Application of appropriate disturbance management regimes such as prescribed burning to reduce fuel loads and reduce the risk of a high-intensity fire.

The various stand-level strategies can often be effectively combined to address a broader range of objectives as part of innovative silvicultural systems that address the twin objectives of commodity production and biodiversity conservation (Taulman et al., 1998; Carey et al., 1999b; Hickey et al., 1999; Beese et al., 2003). For example, the advantages of long rotations are multiplied when accompanied by structural retention at the time of harvest. Conversely, rotation times may be shortened if greater levels of retention characterise logged stands at the time of harvest.

6. Making the checklist operational

Each landscape is unique in terms of physical and biological conditions, human developments (such as roads), the objectives of the landowner(s), regulatory and social directives, and the taxa targeted for conservation. Thus, it is impossible to provide generic solutions to landscape- and stand-level prescriptions for the on-ground implementation of the checklist. As an example in an Australian context, 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 Australian mainland (e.g. in the Central Highlands of Victoria). This is, in part, because of the significant differences in biodiversity (e.g. hollow-dependent vertebrates) between the two regions (Gibbons and Lindenmayer, 2002). 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 be markedly different in another landscape even for the same species (e.g. greater glider [*Petauroides volans*] Lindenmayer, 2002). Forest managers must therefore be quite clear about the objectives they have for a forest landscape and for the stands which comprise that landscape.

It is notable that many current codes and standards of forestry practice usually do not take into account some of the issues and management needs/considerations identified in the checklist such as:

- 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 (Rülcker et al., 1994; Lindenmayer and McCarthy, 2002).
- The importance of maintaining natural disturbance refugia for biodiversity conservation (e.g. multi-aged stands in Victorian ash forests; see Mackey et al., 2002; patches of remnant rainforest in south-east Asia (Johns, 1996)), and the careful management of these areas to ensure their integrity is not impaired by additional human disturbances (Van Nieuwstadt et al., 2001).
- The need to limit multiple cumulative impacts on biodiversity and stand structural complexity in areas subject to forest management (e.g. the combination of logging, fire and grazing) (see McAlpine et al., 2002b).
- The need to ensure that post-disturbance (salvage) logging does not cause negative impacts on biodiversity and forest structure, potentially magnifying effects of wildfires (Shakesby et al., 1996; Lindenmayer et al., 2004; Donato et al., 2006).

7. Other key issues

7.1. *The need for multiple, multi-scaled conservation measures*

Biodiversity is multi-scaled concept. Therefore, attempts to conserve forest biodiversity must also be multi-scaled – with appropriate conservation strategies at the level of individual trees through to landscape and regional levels. 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 landscapes 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). This was demonstrated 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 (Lindenmayer, 2000). This outcome 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).

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

Implementing an array of strategies at different scales 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. Risk-spreading reduces the over-reliance on any one particular conservation strategy and attempts to deal with the considerable uncertainty regarding 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., 2002b), prescribed fire (Moore et al., 2004) and climate change (McCarty, 2001).

7.3. 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 homogenised landscapes. 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 (Franklin et al., 1997). 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).

Different combinations of management strategies or differences in the relative emphasis on particular strategies can sometimes achieve the same objectives for biodiversity conservation. This may allow for trade-offs between such strategies such as: (1) 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; (2) increased levels of stand retention may make it possible to reduce rotation times; and (3) lengthened rotation times with stand retention may allow requirements for strictly protected wide wildlife corridors to be relaxed.

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

Although the general ecological principles and associated checklist presented in this review are intuitive, data on the effectiveness of most specific on-the-ground management actions are limited. Considerable effort is needed to implement trule 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 mitigation strategies and ways to improve practices where necessary. True adaptive management involves rigorous monitoring and a commitment to change when negative impacts of current practices are identified. Unfortunately, the record on forest monitoring (and particularly for biodiversity management) is generally poor in forests around the world (Lindenmayer, 1999) and this needs to be rectified as part of attempts to make transitions to ecologically sustainable forest management.

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