Submission to Legislative Council Select Committee on the Tasmanian Forests Agreement Bill 2012

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Introduction

I make this submission as someone with a deep appreciation for Tasmanian forests and a respect for the many different values that forests represent to the people of Tasmania and the nation. I have a B. Sc. Forestry from the ANU and PhD in forest ecology from the University of British Columbia in Canada. I was born at Ouse and grew up on the northwest coast of Tasmania. I worked for the Tasmanian Forestry Commission for 8 years during the 1980s. I am former Head of Department of Forest and Ecosystem Science and currently Head of Department of Resource Management and Geography at the University of Melbourne and Director of the Victorian Centre for Climate Change Adaptation Research. I have worked in research and policy advisory roles for the Federal Government (including as a member of the team that negotiated the Tasmanian Community Forest Agreement) and in Queensland, Canada, Japan and Papua New Guinea. I have research interests in climate change adaptation, sustainable forest management, the role of forests in providing carbon sequestration and other ecosystem services, forest resource assessment and environmental policy. I am a member of the UN-FAO Advisory Group for the Global Forest Resource Assessment. The views I express are my own.

Key points

1. The structure and activity in the forest industry in Tasmania needs to change. Past policy arrangements that concentrated production and marketing in a single large producer resulted in an industry that was not economically or socially sustainable, or resilient to market or other shocks. A more diversified, innovative and dynamic industry is required that maximises value from smaller logs and (see my attached article published in ABC online). Rates of timber harvesting from public forests should not be locked in. More flexible planning arrangements are required that set harvest levels consistent with broader sustainable forest management objectives (see below).
2. Tasmania's main comparative advantage in terms of wood production is regrowth ash eucalypts. This type of timber is in demand and is not able to be produced in plantations. The future of forest-based industry needs to focus on maximising the value and benefits of this resource to the Tasmanian community. (See attached article by Professor Peter Kanowski, formerly of the ANU).
3. The proposed large scale shift to plantation based production may not be economically or socially sustainable. Eucalypt or acacia plantation forests are energy- and water-intensive, have relatively low biodiversity conservation benefits and are easily established and more productive in other parts of the world that do not have domestic insect pests or diseases. As Kanowski says, extensively-managed self-regenerating native forests, with low inputs and many co-benefits, are a better fit with Tasmania's environment and society. Expanding plantations has also been

controversial. Increased plantations on agricultural land have created strong community reactions due to concerns about loss of community values, farming land, water or aesthetic impacts. Any expansion of the plantation estate will have to be carefully designed to integrate with agricultural land uses and broader landscape-level objectives.

4. The HCV analysis that informed the development of new reserves was hastily undertaken and focused almost exclusively on reserve areas proposed by ENGOs. Consequently, the proposed reserve arrangements do not reflect optimal outcome for dealing with current inadequacies in the conservation reserve system in Tasmania. The proposed reserves focus on vegetation types in 'icon' areas near existing reserves. Many of these vegetation types and growth stages are already well represented in reserves.
5. A more comprehensive and objective assessment is therefore required to inform the development of a soundly-based forest management plan. Otherwise, the pressure will remain to protect further areas, with continued public debate and demands for resources to expand reserves. This assessment needs to consider the longer term implications of climate change and the potential need to shift reserve boundaries as the distribution of species and composition of ecosystems change (see attached paper by Keenan and Read 2012).
6. The way the proposed agreement is framed appears to lock in an 'intensive management or complete protection' paradigm in Tasmanian forest management. Public acceptance of intensive management on more limited areas close to communities has not been tested. The risk is that the public will not like the look of intensive production, scientists will be critical of its impact on other values, and there will be continued calls to stop harvesting.
7. An alternative that needs to be given greater consideration is maintaining and expanding approaches to timber production that provide for high conservation values across the forest estate. This approach would build on the benefits of key reserves and provide for high conservation values and recreation needs across the estate through measures such as variable retention and partial harvesting (based on world-leading research that has been undertaken in Tasmania), thinning around larger trees to promote growth to a minimum size to support the development of hollows, woody debris and other habitat values and providing for the protection of key species within areas managed for timber production (see attached paper by Bauhus et al 2010).
8. This will require appropriate allowances to be made in planning for timber production (more 'headroom'). These objectives can be reflected in the Tasmanian Forest Practices System, which has shown the capacity to provide for a wide range of values in conjunction with timber production. Changes to the process and resources are needed to improve the system, particularly in relation to visual management and design elements of the production system and integration with recreation and other social values.
9. The reduced greenhouse gas benefits of the proposed reserves are highly uncertain and have probably been overstated by some commentators. The potential to generate any market benefits from these is subject to resolution of a range of policy, regulatory and market issues. Assuming a strong short term income flow associated with increased inclusion of forests in reserves is a risky strategy.
10. A final agreement will need to recognise that managing for different values will require ongoing financial investment to maintain access and resources for protection against fire, pests and disease, to effectively manage protected areas for conservation benefits and to provide opportunities for recreation, tourism and production of non-timber forest products such as honey.
11. Ongoing commitment is required to building innovation and R and D and education in the industry. This will support development of new product, improved forest practices and new approaches to forest management for conservation benefits. This does not appear to be recognised in the intergovernmental agreement or the funding package.

The Drum on ABC News 24

25 OCTOBER 2010

A new deal for Tasmanian forests?

ROD KEENAN

The prospect of a resolution to continuing conflicts over forest management in Tasmania is a positive one. However, the process so far has been far from ideal.

A few key parties have been involved in largely secret discussions that have excluded involvement from many people that will be affected by the outcome.

By global standards, Tasmanian native forests are generally well managed. Tasmania has some of the world's highest rates of forest protection in conservation reserves, including 970,000 hectares of old growth forests. Those forest types below current conservation protection targets are largely on private land in eastern Tasmania, where there is little timber harvesting but significant threats from clearing for agriculture or urban development. Environmental NGOs continue to focus on protection for 'icon' forests in south-west Tasmania, but these forest types are relatively well represented in the reserve system.

The biggest sticking point in the agreement is likely to be the proposed transition of the forest industry out of public native forests into plantations. There are currently insufficient plantations to meet wood supply commitments or replace the level of industry activity and employment from native forests. Only 10 per cent of current Tasmanian plantations can produce higher value products. The current crop of eucalypt plantations was established largely for pulpwood, either for export or for use in the proposed pulp mill. They are generally not of the right species or varieties, nor have they been managed to produce products for construction, flooring or joinery.

Consequently, it will take some time to establish a sufficient area to a replacement resource. It will take 20 to 40 years (depending on site and management) for new plantations to provide higher-value products. Further research will be required to support their management and new types of processing will be required to produce higher value products.

Expanding plantations has also been controversial. Increased plantations on agricultural land have created strong community reactions due to concerns about loss of community values, farming land, water or aesthetic impacts.

Consequently, developing an increased plantation estate will take time to build community support and the knowledge base for plantation production. A similar deal struck between government, industry and conservation groups in south-east Queensland about 10 years ago has not yet resulted in sufficient public or private investment in new plantations to offset losses in timber production from native forests.

It will also take money. Given the failure of most companies involved in plantation-based managed investment schemes and the controversy surrounding the sector, there is little interest in banks or the finance sector in investing in new plantations. Public funding will necessarily need to come from the Federal Government, with arguments for this investment competing with water buybacks, irrigation, infrastructure, education or health commitments.

New investment models will be required to provide the necessary finance. Some are pointing to financing from climate change and carbon. While the carbon benefits of new plantations are clear, the potential benefits of phasing out native forest harvesting, in the face of increasing fire risks and other impacts of climate change are highly uncertain.

The need for the Tasmanian agreement has been driven primarily by changes in international timber markets. Australians use the equivalent of 22 million cubic metres of wood each year in timber, paper and other forest products. Thirty per cent of this currently comes from native forests. With our high value dollar, the forest sector in Australia needs to go high-tech.

In Europe the forest sector and government are investing heavily in research to support new types of engineered and laminated wood products, biochemicals to replace petrochemicals and in clean, highly efficient, wood-based bioenergy systems. We need research and industry investment to maximise volume production, value recovery and resource use efficiency from the wood we do harvest.

These new technologies will require more highly skilled and trained professionals and technicians that can bring wider benefits to Tasmania and other parts of regional Australia. However, investment in research, from industry or government, requires assurance of resource security and a long-term future for the industry.

While the agreement is focused on protecting 'high conservation value' forests, defining these forests has proven challenging. In my view, we should be providing for high conservation values across our forest estate. All forests, including those managed for timber production, should be managed to provide clean water, biodiversity, carbon, soil protection, recreation and pollination benefits in a multi-functional, landscape-scale approach. This philosophy is widely promoted internationally, but in Australia we seem to be stuck in a limited vision, 'ecological apartheid' model, where conservation and production must be clearly segregated.

We might overcome this by improving the aesthetics of forest operations. Foresters have developed management practices that result in effective regeneration and retain most landscape-level biodiversity, but, let's face it, they often look pretty bad. We are now more demanding in visual and functional design in the built environment. We need to adopt the same principles in managing our natural environment.

I have been a close observer of forests and forestry in Tasmania all my life. The forest sector in Australia is in a turbulent period. This can provide the opportunity for creativity and innovation to drive new models of forest management, new products and new industries. A lasting and sustainable agreement on forest management in Tasmania will require new thinking and some tough choices. It is only likely to be achieved through genuine, long-term engagement in decision making from all parts of the Tasmanian community. With that, Tasmanian forests may well become, in the words of Dr Kerry Arabena, co-chair of the Congress of Australia's First Peoples, "landscapes of reconciliation".

Rod Keenan is a Professor of Forest and Ecosystem Science at The University of Melbourne. He grew up in Tasmania and worked there in forest management and research for 10 years.

THE CONVERSATION

7 June 2011, 10.58am AEST

Pulping Tasmania's future

Peter Kanowski
Professor of Forestry at Australian National University

The peace talks underway about Tasmania's forests are as rich in ironies and paradoxes as Tasmania's old-growth forests are in carbon.

Who wouldn't walk away from an agreement that locks Tasmania into a backward future? ialla/Flickr

The current direction of the peace talks locks Tasmania into a pulpwood future, the very situation critics of Tasmanian forestry have been arguing against since woodchipping started in the 1970s.

This future ignores Tasmania's comparative advantages in forestry and the bigger global picture. It delivers marginal carbon and biodiversity gains at unnecessary economic and social cost.

There are certainly elements of the peace deal that should endure, but the current package throws the forests baby out with the proposed pulpmill effluent water.

Plantations are good for pulpmills

The first irony is that critics of Tasmanian forestry have long argued that the state's forest and industry policies gave too much weight to the interests of one dominant company, Gunns Ltd.

The peace deal continues that tradition. It disproportionately reflects the interests of one corporation, not those of the forestry sector or the community more broadly.

The second irony is related. Tasmania's Regional Forest Agreement, signed between the Australian and Tasmanian Governments in 1997, traded off an increase in national parks against an increase in turning other native forests into plantations.

Many environment groups, with a narrow focus on protecting old-growth forests, effectively acquiesced. 150,000 ha of native forests valuable for biodiversity, carbon, and wood were converted to plantations before December 2007, when the practice stopped.

WWF, in its 2004 Blueprint for Tasmania's Forests, was one of the few to point out that this was a bad outcome. But the expanded plantation area forms part of the resource that allow

Gunns to propose the plantation-only pulpmill that is associated with the peace deal.

A good process doesn't happen behind closed doors

A third irony is that the peace process has breached the principles of good forest governance. Environment groups have argued persuasively that these should be the foundation of Tasmanian forest policy and management.

Those principles include inclusivity and transparency. Both of these are difficult to achieve in invitation-only closed-door talks convened by the two parties with the most extreme interests – no logging, on the one side, and logging on the other.

The forest agreement process needs more voices. (AAP)

Those with positions that don't align with those interests, and who might see the situation in rather less black-and-white terms, have no voice.

Reserving forests won't solve our problems

Protagonists have found common ground by excluding or silencing those with other views. They are seeking substantial public funding to solve a concocted problem. Tasmania's forests are not, in fact, under any imminent threat from which they need to be "saved".

The carbon emissions associated with harvesting all of Australia's native forests form a trivial proportion – a few percent – of national greenhouse gas emissions.

The overwhelming majority of forest-related emissions and biodiversity loss are associated with clearing forests to make way for farms, houses and plantations. (Over the decade to 2008, 90% of this clearing was for agriculture and urban development, and 10% for plantation conversion – another double whammy).

The paradox is that Tasmania does have a global comparative advantage in growing native forest timber.

The global comparative advantage in *plantation* production is in South America, where the growth rates of eucalypts in pest-free exotic environments and the scale of plantation development deliver extraordinary production advantages.

Indonesia's advantage is that deforestation associated with forest products appears of little concern to Australians who consume them. There, plantation forestry generates the stinging critiques we usually associate with Tasmanian *native* forest politics.

Get the trees out of the woods

There is a bigger global picture, which received a little national airing before the global financial crisis intervened. There is a looming crises in global food, energy and water supplies. With the added impact of climate change, we will have to change the way we manage rural landscapes to survive these.

These issues are rightly the focus of growing concern and strategising globally, but have so far received scant attention in Australia's peculiar electorally- rather than policy-focused contemporary politics.

The broader international consensus is that we need to transition to carbon- and energy-positive landscapes. These must use less water for food and fibre production than do current systems.

They must maintain biodiversity across the landscape rather than just in reserves. They must be resilient to climate change. They need to spread rather than concentrate risks.

Saving forests isn't enough: we need more trees on farms.
(Jane Rawson)

One solution: forested landscapes which are well, but not completely, reserved, and farming landscapes with more trees.

Plantation forests, as relatively energy- and water-intensive, and biodiversity-poor, production systems, have only a partial role in such a future.

Extensively-managed self-regenerating native forests, with low inputs and many co-benefits, are a better fit. So are other forms of tree growing more integrated with agriculture.

Ironically, Tasmania is well down some parts of this path – a third of its native forests are already reserved, there is strong focus on conservation of private as well as public forests, and it has a forest practices system that scores highly in global comparisons.

However, like the rest of Australia, it needs coherent and sustained public policy supporting integrated and sustainable management of predominantly agricultural landscapes.

A curate's egg: good in parts

The peace deal on the table has elements that the Australian and Tasmanian Governments should support: an end to old growth harvesting, reducing the volume of sawlogs that Forestry Tasmania is legally required to deliver, and ample exit packages for those whose employment depended on an earlier era of native forest harvesting.

But there’s no need to fund a transition away from harvesting native forests for high-value wood products. Tasmania has a natural advantage in this industry.

What’s needed is a harvesting regime where biodiversity and carbon stocks are protected in an adequate reserve system and valued by the market. This, in turn, requires a price on carbon.

Rather than buying out businesses that don’t need to close, the Australian and Tasmanian Governments would to better to direct public funding to developing and supporting integrated production systems that address the real issues in sustainable management of Australia’s landscapes.

These are in our agricultural, not our forested, landscapes, both in Tasmania and mainland Australia. That’s where building a lasting peace most needs our attention and public funding.

Helping develop Tasmania as a global showcase for that environmentally-friendly future would be a public investment worth making.

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Assessment and management of old-growth forests in south eastern Australia

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OLD GROWTH FORESTS

Assessment and management of old-growth forests in south eastern Australia

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Abstract

Old-growth forests in south eastern Australia are important for biodiversity conservation, recreation, carbon storage, social values and, to a declining extent, for timber production. Developing a comprehensive definition of old-growth forest that can apply across all Australian vegetation types has been challenging. Old growth can be viewed from ecological and social perspectives. For policy and management purposes old growth has been defined as a growth stage in forest development, incorporating ecological maturity and lack of evidence of past disturbance. Classification and assessment of old growth has largely been restricted to those areas covered by regional forest agreements (RFAs) between different states and the Federal Government. Old growth can be impacted by wildfire, timber harvesting, insect pests, diseases and other disturbances. Climate change will also present challenges for the future management of old-growth forests. There is increasing scientific understanding of the relationships between species, forest growth stage and old-growth forest attributes. To meet biodiversity conservation objectives, the management focus is shifting from assessing and protecting old-growth forests, to providing for forests across the landscape with old-growth attributes. This approach may be at odds with other conceptions of old growth based on notions of undisturbed systems free of human influence.

Keywords: *Ecology, climate change, old-growth, dynamics, forest management*

Introduction

Primary, or old growth, natural forests are important global assets (Franklin et al. 1981; Beese 2003; Beadle et al. 2009; Marchetti et al. 2010) that are subject to continuing loss (FAO 2010). Australia has about 149 M ha of natural forests (Montreal Process Implementation Group for Australia (MPIGA) 2008) of which 3.4% have been formally mapped and described as old growth for forest management purposes. Forests in Australia range from dense, tall forests dominated by eucalypt or rainforest species to open short woodland and mallee formations. Old-growth forests have been a significant part of forest management and planning in Australia since the 1980s. Old-growth forests are considered significant because they have habitat, nature conservation and aesthetic values that are not found in other forests.

Australian forests have had a long history of human use and disturbance, initially by indigenous peoples, whose impact on forests was primarily through the use of fire, and later by European settlers, through conversion to agriculture and timber harvesting and the displacement of aborigines and their burning from most landscapes. Forest use was largely uncontrolled until the early 1900s, when concerns over timber resource depletion, loss of water quality and growing demand for recreation from an expanding urban population led to the establishment of state forest management agencies, more systematic licensing and control of forest operations, and the development of scientifically-based regeneration and other silvicultural practices. Forest resource use became more intensive in the second half of the twentieth century, with conversion to plantations in some places, and the introduction of extensive harvesting of native forests and associated industries to utilize residual wood in papermaking or woodchip exports.

This article reviews concepts and definitions of old-growth forests that have been applied in south

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eastern Australia. We provide an analysis of the current status of old-growth forests across different types of land tenures and discuss the impacts of disturbance, including timber harvesting, on old-growth forests and the potential consequences of climate change. We consider future management options for old growth with a focus on the maintenance of old-growth forest values across large forest estates with multiple tenures and forest types.

Old growth concepts

"Old growth" emerged as a widely used term in Australia in the 1980s. Old growth integrated elements of the relatively old concept of "wilderness", with the emerging understanding of biological diversity that was being codified through the discipline of conservation biology. "Old growth" also contained connotations of maturity, venerable age, primitive origins or lack of disturbance by modern technology (Beadle et al. 2009).

In the 1980s, different utilitarian, conservation and aesthetic world views collided at a time when rapid social and environmental change was leading many to question the underlying rationale for societal development, economic growth and the meaning of "prosperity". A growing environmental movement driven by concerns of widespread pollution and loss of natural areas and species became an increasingly powerful political force that drove significant change in forest and natural resource policies. Forest managers, generally trained as rational agents of the state or corporation, were poorly prepared for this social change (Lee 2009).

In Australia, this collision initially focused on protection of rainforests. Rainforests in Australia extend from the tropics to cool-temperate regions. They are defined by a suite of species with particular ecological requirements, including a closed canopy cover (>80%), generally high year-round rainfall and the capacity of many species to regenerate below a closed canopy (Bowman 2000). Most rainforests contain no eucalypts. In tropical regions rainforests are characterized by high species diversity and structural complexity while there are relatively few tree species and less structural complexity in temperate rainforests. In high rainfall situations and in the absence of disturbance, old-growth eucalypt forest will transition in a successional sequence through "mixed forest" to rainforest. In some situations, this can happen relatively and rapidly (<100 years) as the development a rainforest understorey can accelerate the death of the eucalypts (Close et al. 2009). Rainforests are currently found in only 2% (3.2 M ha) of the continent and, while they have been heavily impacted by land clearing and timber harvesting, are now generally well-protected with 55% of the current area in conservation reserves (MPIGA 2008).

Other concerns about forest management practices, including conversion to exotic pine plantations and more intensive management associated with harvesting of pulpwood for export, also became more important during this time. There are over 730 eucalypt species in Australia and these dominate on 78% of the total forest area. Eucalypts occur in a range of conditions from high-rainfall tropical and temperate areas to semi-arid zones. They generally depend on some form of disturbance to regenerate and are capable of persisting after fires that occur with varying frequency and intensity depending on site conditions and ignition sources. Timber production has generally concentrated in the tall, open forests in wetter conditions in eastern and south-western Australia.

Old-growth eucalypt forests provided a focus for the attention of an increasingly vocal environmental movement buoyed by their success in rainforest protection and stopping construction of a dam that would inundate the Franklin River in Tasmania. The term had come into widespread use in the USA in the 1980s, to describe forests with tall, large-diameter trees in coastal rainforests in Oregon and Washington (Franklin et al. 1981). In Australia, eucalypt forests with similar canopy structures to these old-growth coniferous forests can be found in cooler, wetter parts of the country where disturbances such as fire occur but are relatively infrequent.

A complex mix of politics and ideological differences between different state and federal governments also led to differences in views over forest management. This led to a federally-initiated Resource Assessment Commission Inquiry into forests (Resource Assessment Commission (RAC) 1992), and development of a National Forest Policy Statement (NFPS) that aimed to provide the basis for an agreed path for forest conservation and development. The Commission undertook an analysis of the status and use of old-growth forests. The NFPS provided a nationally-agreed definition and statement of old-growth forest values.

A number of ways of "framing" old-growth emerged during public debate at this time:

(1) Old growth as a concept based in ecological science, describing a stage of development of a forest stand or ecosystem.
(2) Old growth as a broader concept describing large, intact areas where natural disturbances and processes dominate and where forests may have the potential to become old growth.
(3) Old growth as a social construct-based around forest "age" and lack of disturbance by humans.

The first of these frames recognised that forests are dynamic systems that go through stages of development following disturbances such as fire, windstorms or timber harvesting. However, disturbance events have a range of severity. They are not always completely stand-replacing and some trees and legacy structures can survive even severe disturbance. Thus, even forests subject to recent disturbance may have "old-growth attributes" (Bauhus et al. 2009).

The second recognised that a mix of vegetation growth stages occurs across larger landscapes and provides for habitat or other ecological processes, and that the pattern and structure of vegetation in the landscape is dynamic (Burgman 1996). The important characteristic is that any part of the landscape should have the capacity to develop into a mature or old-growth stage at some time in the future.

In the third framing, old-growth forests are socially-constructed icons, with large, old trees likened to the "charismatic megaflora" of conservation (Kanowski and Williams 2009) and systems undisturbed by humans providing a sense of refuge from the modern, technological world or places that have a right to exist, uninterfered with by humans. Forests with these elements, that dwarf humans in physical stature and lifespan, have a powerful emotional impact and people hold strong views about relatively rare resources that take a long time to develop (Dovers 2003). These elements became effectively used as symbols in debate over native forests use.

Identifying the defining characteristics of old growth under this last conception is difficult. Assessing and representing the "imagined" values of forests in terms compatible with rational frameworks such as criteria and indicators of sustainable forest management that emphasize quantification and objectivity is a significant management challenge (Kanowski & Williams 2009).

Defining and mapping old-growth forests in Australia

The 1992 National Forest Policy Statement (NFPS, Commonwealth of Australia 1992) made specific provision for the protection of old-growth forests. The NFPS provided for a process for undertaking assessments of forests for conservation values, including old-growth values. This was based on the following definition (based in the "stage of stand development" framing, discussed above):

> ...forest that is ecologically mature and has been subjected to negligible unnatural disturbance [...] in which the upper stratum or overstorey is in the late mature to overmature growth phases.

A working group of state and Australian Government agencies took the NFPS definition into consideration in developing a definition that was accepted by state and federal governments (JANIS 1997).

> Old-growth forest is ecologically mature forest where the effects of disturbances are now negligible.

Ecologically maturity is a key feature. This includes the presence of trees that would be expected in such a forest type in a condition consistent with a lack of large-scale disturbances for a long period. This includes trees in an over-mature or senescent growth phase that are no longer actively growing, or that may be reducing in size due to crown dieback and branch shedding.

Workshops with ecological specialists identified specific attributes that could be used to characterize and map old-growth forests (e.g. Dyne 1992; Love et al. 1993). Forests in the old-growth stage have a layered structure with large overstorey trees, a well-developed understorey of other tree species, shrubs and ecological features such as dead standing trees and large logs on the forest floor. Features such as hollows in which fauna can nest are usually also more prominent. These are generally consistent with the suite of attributes associated with old growth in other parts of the world (Bauhus et al. 2009; Table 2) except that eucalypt-dominated old-growth forests do not generally have understories dominated by late-successional or old-growth species, high levels of advanced growth of the canopy species or thick forest floors. Where multiple age cohorts are present, these are usually limited in number and associated with larger or smaller-scale disturbances, not continuous recruitment. The presence of multiple age cohorts in old-growth forests varies between regions and between forests dominated by different eucalypt species (Turner et al. 2009).

Early analysis of the defining attributes of old-growth forests in Australia did not have a specific focus on biomass density and carbon stocks. This has become more important in forest policy recent years and there have been a number of studies demonstrating that old-growth eucalypt forests in temperate regions can contain high carbon density compared to many other forest types or growth stages (Raison et al. 2003; Dean & Wardell-Johnson 2010; Keith et al. 2009, 2010).

Further development of this definition enabled consistent application across different forest types (Pitman et al. 1996). Forest composition and structure vary considerably across Australia, and therefore different states adopted different definitions for assessment of old-growth forest values in Comprehensive Regional Assessments undertaken for regional forest agreements (RFAs) between state and federal governments in four states. These varying definitions reflected differences in forest type between states and provided a basis for mapping old growth from aerial photographs.

Applying a general definition of old growth across forest types has challenges, with Lindenmayer (2009) arguing that a scientifically defensible and ecologically robust definition only becomes valid when applied to a particular forest type within a given region. Applying the old-growth concept in drier regions with frequent fire is also a challenge. In these forest types, older forests are often structurally less diverse, with a canopy of large, older trees (often with hollows), a sparse understorey and ground cover of native grasses. Overstorey species generally do not develop the same crown characteristics found in tall eucalypt forests in wetter areas.

Old-growth forest assessment and extent

Different approaches to definition and mapping were used in different Australian states (Keenan & Ryan 2004). The JANIS definition was generally adopted, but in Tasmania a detailed rule set was developed for different forest types based on crown senescence characteristics and levels of disturbance (Tasmanian Public Land Use Commission 1996). In Victoria, old growth was forest containing significant amounts of the oldest growth stage (usually senescing trees) in the upper stratum. For mapping purposes, a maximum regrowth crown cover of 10% was allowed, as areas with regrowth crown cover of more than 10% are almost always associated with significant unnatural disturbance. The JANIS definition was also adopted in New South Wales, but a maximum regrowth crown cover of 30% was used to define senescing forests.

Mapping old-growth forests requires knowledge of growth stage and disturbance history. Disturbance often cannot be easily characterized through remote sensing or aerial photograph interpretation, and on-ground assessment is often required to assess forest structure and evidence of tracks, stumps and fire scars (MPIGA 2008).

A mix of growth stages is likely to be present in most Australian forests as a result of previous disturbances. The total area for which the growth stage of the forest is known is almost 15.4 M ha. Growth stage has been mapped on State Forests but not generally on private land or conservation reserves. Non-eucalypt communities, such as rainforest or drier open acacia woodlands, also cannot easily be classified by growth stage.

About two-thirds of the area with known growth stage (10 M ha) is classed as mature or senescent, and a total of 5.03 M ha of old-growth forest was identified in the RFA regions in 2008. This was about 200,000 ha less than that reported in 2003 due to the impact of severe fires, which converted some areas of old-growth forest into younger age classes, and some remapping (Table I, MPIGA 2008). Almost half of Australia's total identified old-growth forest is in New South Wales, and most of it is on public land. The proportion of the forest estate that is old-growth forest varies widely by state. In Tasmania almost 40% of the forest cover is old growth. Examples of maps of old-growth forests in these regions can be found at http://www.daff.gov.au/rfa (Accessed April 2011).

Old-growth forest conservation status

For the RFAs, conservation targets were set to provide for sufficient representation of all forest communities in reserves, protection of high quality wilderness areas and provision for the protection of rare or depleted habitats and species. The following reservation targets were agreed following a process of consultation between state and federal governments as follows:

(1) For those forest types where old-growth forest is rare or depleted (defined as less than 10% of the current distribution of its forest type), all viable examples should be protected (100% target)
(2) For other forest types, 60% of old-growth forest should be protected, with appropriate flexibility in the target applied to ensure:

Table I. Area (,000s hectares) of old-growth forest in regions assessed for Regional Forest Agreements.

State	Native forest area	Mapped old-growth	Percent	Old-growth on public land	Old-growth on private land	Old-growth in conservation reserves	Percent
NSW	8989	2536	28	1892	644	1742	69
Qld	3230	270	8	196	71	196	73
Tas	3116	1228	39	1118	110	973	79
Vic	5774	673	12	673	1	460	68
WA	1909	331	17	331	n/a	331	100
Total	23,018	5039	22	4209	826	3702	73

Note: From MPIGA (2008).

- old-growth forest representation is from across its range;
- high-quality habitat areas are included;
- reserve design is appropriate (i.e. can be managed practically);
- largest and least fragmented areas are protected; and
- community needs for recreation and tourism are met.

As a result of the RFAs, a more recent agreement between the Federal Government and Tasmania, and other state government policies, about 73% (3.7 M ha) of current old-growth forests are now in formal or informal conservation reserves. While the 60% minimum target by vegetation type has been met in some states (e.g. all old growth on public land is now in formal or informal reserves in southeast Queensland and Western Australia) there are some forest types that remain below this threshold. These were either old-growth forest areas on public land deemed to be required to meet social and economic objectives specified in the RFAs, or vegetation types that occur predominantly on private land and where the conservation targets could only be achieved through land purchase or conservation agreements with private landowners (both primarily in Tasmania).

Old-growth forests occur on both public and private tenures in Tasmania, northern NSW and South East Queensland. The pattern of old-growth forest distribution varies considerably between tenures. The majority of large, intact areas of old growth are in conservation reserves and the extent and patch size of defined old-growth areas in State Forests available for timber production are relatively small. Patches on private land are smaller still, due to past harvesting or grazing disturbance. About 26% (1.6 M ha) of the conservation estate with known age class is not classed as mature or senescent (MPGIA 2008).

Impacts on old-growth forests

While old-growth forests can take a long time to develop, they can also be lost quite quickly under catastrophic disturbance such as wildfire. Thus, for eucalypt forests (and other forests that experience significant disturbance) the conception of old-growth forests as durable and immutable is inconsistent with ecological reality. Tall, wet, iconic eucalypt forests are fire-dependent, requiring disturbance in the form of a particular fire regime to reproduce and persist. A single fire may not significantly impact on old-growth characteristics, with some large live trees often remaining after fire and dead standing trees and downed logs contributing to old-growth attributes. However, absence of fire over the long-term will lead to successional development and replacement by non-eucalypt forest types such as rainforest. With more frequent fire, old-growth eucalypts can be replaced by different species of lower stature, different understorey characteristics and habitat values.

Old-growth and mature forests historically provided the primary resource for the timber industry in some parts of Australia, especially for high-quality sawn timber and veneer. Old growth is lost when areas are harvested and regenerated, whether selection or clearfelling methods are used, although some old-growth "attributes" can remain, or be retained. Old-growth stands were initially logged and left, with little consideration given to regeneration, but by the 1960s regeneration practices had been developed for most forest types. Industry reliance on mature or old-growth forests has now declined to a low level in most states. There is little premium for larger logs, except in Tasmania, and the native hardwood industry is now generally geared to harvesting and processing smaller logs from regrowth forests.

The inclusion of old-growth forests in reserves has had significant impact on wood supply. For example, in Western Australia, sawlog supply declined from 457,000 m^3 per year to 185,000 m^3 per year partly as a result of old-growth protection policies, because some regrowth or mature forests also being unavailable as a result of the new reserve boundaries.

High-intensity fires are significant threats to old-growth forests across all tenures. Considerable areas of old-growth forests in southeastern Australia have been burnt in wildfires in recent years (mostly in 2003, 2006–2007 and 2009) and structurally they are now dominated by trees in earlier growth stages, although significant old-growth elements can remain, including large standing dead trees and woody debris. Diseases such as the *Phytophthora* root-rot can also impact on both old-growth and regrowth forest growth stages.

In practice, the application of social constructs and definitions based on values of forest "age" or presumed naturalness often leads to preference for high proportions of old-growth forest in the landscape, coupled with fire suppression. However, this can lead to higher fuel loads, elevated fire risk, and more intense and uncontrollable fires (Covington et al. 1997; Binkley et al. 2007). Given the ubiquity of fire disturbance in much of the Australian landscape, it may be necessary to provide situations where old-growth stands may develop in the future. Lindenmayer (2009) argues that regrowth areas be set aside to eventually develop into old growth and provide for temporal changes in stand structure, uncertain environmental events, and the spatial processes and dispersal mechanisms of old growth-dependent flora and fauna in places where old growth is scarce. Ferguson (2009) similarly suggests that a cross-tenure, whole-of-landscape perspective be adopted in maintaining old-growth values in the Australian tall, wet ash-type eucalypt forests.

Old-growth forest monitoring, assessment and management

As discussed, the three conceptions of old growth represent challenges for forest planners and managers. Many in the environmental community do not accept the relatively prescriptive ecological definition of old-growth forests based on forest growth stage. They would prefer monitoring and management to include larger-scale, landscape-level aesthetic values and ecosystem processes, whether or not these are encompassed in any formal definition of "old growth". In Australia, assessment and conservation targets have generally been determined at the large regional scale (greater than 1 M ha) but the scale of assessment and mapping old growth has been at the scale of small patches (down to 2–3 ha). There have been few studies of the dynamics of forest growth stages across large landscapes in Australia. As Spies (2009) points out, the extent and type of old growth in a landscape or region can fluctuate considerably over time with natural disturbances. Establishing the "right" proportion and distribution of old growth in a particular jurisdiction is a social and political issue rather than a scientific one.

The threshold extent of human disturbance at which forest is defined as old growth has been a challenge for old growth assessment. While evidence of recent human intervention may be important in wilderness-oriented conceptions of old growth, it is less important for conservation biology. While some wildlife species (e.g. in Victoria Yellow-bellied Glider, *Petauus australis* and Sooty Owl, *Tyto tenebricosa*) are likely to be dependent on large, intact areas of old-growth eucalypt forests, many native bird and mammal species are most abundant in old growth stands but are not confined exclusively to them (e.g. the Greater Glider, *Petauroides volans*, is more dependent on old-growth forest attributes (dead trees, hollows or downed woody debris) rather than on lack of disturbance *per se*). Increasing these elements in regrowth forests can make an important contribution to biodiversity conservation (Lindenmayer & Franklin 2002; Lindenmayer 2009). Old-growth forest understorey plants may also exist within a regrowth forest and these plants (such as old tree ferns) can be important places for the establishment of epiphytic elements (Mueck et al. 1996; Lindenmayer 2009).

A number of species are reliant on forests containing old-growth forest attributes, because of the range of nesting hollows and greater structural complexity they have in comparison with forests in earlier stages of development. However, many vertebrate species require the presence of more than one growth stage for survival (Lindenmayer 2009).

Consequently, if conserving old growth-dependent biodiversity is the goal, maintaining old-growth attributes across a forest estate may be a better management option (Bauhus et al. 2009). Many of these are not generally being provided in current management systems for timber production (Table II). If new management objectives are adopted this will cause a shift from aiming to meet given area targets for protection or percentage of forest cover in an old growth stage, to setting targets for structural or compositional features such as dead, large and old trees, varied stand structure, canopy gaps, woody debris, old understorey species elements and forest floor components. This will require a shift in monitoring approach to one that identifies the extent of these attributes across the estate. Assessing these old-growth forest attributes is likely to require new approaches to forest assessment (e.g. Corona et al. 2010), and new developments in remote sensing are improving capacity to assess old-growth forest characteristics (Ohmann et al. 2007).

Managing for "old-growthness" can involve both maintenance of existing stand conditions and active intervention to increase development of features and stand structures to support old-growth forest species or characteristics (Table II). This can include thinning, patch burning or damage to trees to encourage hollow development. These interventions can yield products for sale and may provide for a combination of uses in situations where conservation and timber production are desired social values from natural forests (Bauhus 2009).

"Variable-retention" approaches are being applied in harvesting in old-growth forests in Tasmania (Forestry Tasmania 2009), where there is a high proportion of forest as old-growth, using similar approaches to those in north America (Beese et al. 2003). This system explicitly considers site variability and particular features that merit retention and the natural disturbance regime applying to a particular forest. Unharvested aggregates are retained in tall, wet old-growth eucalypt forest. These are significant intact patches of old-growth trees and other late-successional species, and provide sources of seed, spores or fauna that can establish in nearby regrowth forest. These types of harvest systems may be more similar to natural disturbances regimes that result in multi-cohort stands (Turner et al. 2009). Other areas are being managed over longer rotations to provide for a mix of specialty timber species and other values.

In Victoria, where most of the forest area harvested is regrowth, retention of patches in harvested areas

Table II. Old-growth forest attributes (Bauhus et al. 2009) and their presence in old-growth wet eucalypt forests, in ecosystems managed for wood production and capacity for provision through alternative management approaches.

Old-growth attribute	Present in Australian old-growth eucalypt forests	Present in currently managed eucalypt forest ecosystems	Capacity to be provided in managed eucalypt forest ecosystems
High number/basal area of large trees	Yes	No	In some management regimes
High stand volume or biomass	Yes	Yes, for oldest stands	Possibly
Large number/basal area of dead/dying standing trees	Yes	No	Possibly
Large amount/mass of downed CWD	Yes	Yes but less in future rotations	Yes
Wide decay-class distribution of logs and/or snags	Yes	Yes	Yes
Several canopy layers/vertical variability	Yes	In some forest types	Yes
High number/cover of late-successional/ shade-tolerant species	Generally No	No	No
High variation in tree sizes, presence of several cohorts	In some forest types	In some forest types	In some management regimes
High spatial heterogeneity of tree distribution/ irregular size and distribution of gaps	Yes	Yes	In some management regimes
Thick forest floor	Not usually	No	No
Special attributes (pit and mound relief, presence of epiphytes, presence of cavity-trees, tree hollows)	Yes for epiphytes and hollows	Limited	Yes
High variation in branch systems and crown structure/development of secondary crowns	Yes	Not generally	Yes
Presence of advance regeneration	Not usually	In some forest types	Yes

that can develop into future old-growth areas is being tested (Lindenmayer 2008). Targeted thinning is also being considered that can promote stand development and the ageing of regrowth, mimicking the effect of drought, insect attack or low-intensity fire by killing or removing some individual trees, increasing ground-level coarse wood debris, and promoting structural complexity (Hamilton 2009).

Increased management complexity and lost timber production potential means that implementing this kind of management generally costs more than traditional silvicultural approaches. While it may be feasible technically to retain and restore complex forest structures, funding these approaches is a major challenge. In the case of Tasmania, the Federal and State Governments jointly contributed A$250 million to a package of activities to provide for protection of a further 125,000 ha of old-growth forest in reserves and research and development of new management approaches for old-growth (TCFA 2005). In Victoria, government programmes have been implemented to encourage private owners to forego harvesting, or to restore and protect native vegetation.

These practices are currently being used on a limited extent of the forest estate. In Tasmania 750 ha of old-growth forests each year are being harvested using retention systems, divided approximately equally between wetter and drier forest types.

Establishing who bears the costs (in either the public or private sector) for different types of silvicultural practices in multiple-use forests, particularly where there might be foregone opportunity costs, is a policy challenge (Bauhus et al. 2009). In Tasmania, retention practices are being implemented in order to support a wider "social licence" to operate in natural forests. Implementation of these practices on a larger scale in Victorian conservation reserves is dependent on public funds. In the case of State Forests the commercial management group (VicForests) needs to be willing to forego potential revenue. Incorporation of requirements for these types of practices in forest certification schemes could become a market incentive for maintaining old-growthness on some part of the managed landscape (Bauhus et al. 2009).

Climate change and old-growth forests

In SE Australia, climate will become drier, seasonal rainfall patterns will change and the frequency of high-intensity wildfires is forecast to increase (Hennessy et al. 2005). This will have implications for the structure and condition of current old-growth forests and their capacity to regenerate. As discussed, moderate fire disturbance may enhance conservation values, but large-scale intense wildfires whether natural or human-induced may result in the loss of the undisturbed and aesthetic appeal that people commonly associate with old-growth forests.

Maintaining old-growth forests in the face of a rapidly changing climate will require an improved understanding of how climate change might affect rates of growth, stand development, mortality and regeneration and how species distribution patterns might shift in response to climate change. Climate change might accelerate development of old-growth features in the short-term, through increased growth, senescence and mortality. In the longer term, species shifts, changing fire patterns and increased incidence of pests and diseases might see the displacement of iconic tall, wet forest ecosystems with those more able to persist in drier conditions with more frequent disturbance, or with grass or shrub dominated systems. Ferguson (2009) highlighted the need to plan for adequate seed collection and storage, and the use of artificial regeneration, to maintain tall, wet ash forest types in Victoria in the face of climate change.

Arguments have been presented to halt timber harvesting in old-growth forests on the basis of potential carbon emissions from these practices (Mackey et al. 2008; Dean and Wardell-Johnson 2010). This would require changes in Australia's current policies on inclusion of ''forest management'' towards greenhouse gas emission reduction targets, improved measurement and accounting to establish baseline levels of emissions, analysis of the economic costs and social implications compared to other options for reducing greenhouse gas emissions, and analysis of the loss of carbon stocks in old growth from fire, and increased risks associated with climate change (as discussed above).

In managing forests for old-growth values in the future, forest managers will have to resolve significant questions with the community, including the degree of intervention and human disturbance that is acceptable in native forests to promote development of old-growth features, the place of roads for access and protection, and the use of prescribed burning and intervention to manage pests and diseases in natural systems. In adapting management of old-growth forests to climate change, forest managers will have to consider the acceptable degree of intervention that is appropriate to maintain health, condition or existing species composition, and whether translocation or regeneration with genetic material is appropriate to maintain species composition or conservation values.

Conclusions

Old-growth forests have been the focal point for public debate over the management of natural forests in Australia for the last 20 years. Significant effort has gone into defining and mapping old growth in south eastern Australia, and protection of old-growth forests has been a dominant element in development of a comprehensive, adequate and representative reserve system.

New management approaches are being developed and applied for old-growth forests, particularly tall, wet forests that have traditionally been harvested and converted to regrowth with silvicultural systems involving clearfelling, burning and sowing. These new management systems aim to maintain old-growth attributes within harvested stands, and provide for conservation of flora and fauna at both the stand and the landscape level. Management systems are also being trialled to provide greater levels of ''old-growth attributes'' in regrowth forests and other vegetation types.

These new management systems might satisfy those that are primarily concerned with biodiversity conservation but are unlikely to satisfy those concerned with perceived loss of a broader set of aesthetic, spiritual or cultural values from activities such as timber harvesting. A number of environmental groups have policies that harvesting in all natural forests in Australia should cease. Public discourse in forest debate is shifting from protection of old growth to other terms, such as ''high conservation value'' that encompass old-growth and other forest growth stages with different values (e.g. Forest Stewardship Council (FSC) 2010). However, similar effort has not currently gone into defining and mapping these high conservation value forests as has gone into defining and mapping old growth.

Providing for a wider range of products, services, carbon and biodiversity habitat values (including old-growth values) in managed, multi-functional forest landscapes may meet conservation biology objectives, but it may not satisfy other political needs driven by the desire to protect forests considered to be ''wild'' or ''untouched''. Further research is required to develop approaches that can effectively integrate a wider range of values (aesthetic, spiritual or cultural) into monitoring and management frameworks.

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Silviculture for old-growth attributes

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ABSTRACT

Silviculture to maintain old-growth forest attributes appears to be an oxymoron since the late developmental phases of forest dynamics, described by the term old-growth, represent forests that have not experienced human intervention or timber removal for a long time. In the past, silvicultural systems applied to old-growth aimed to convert it into simplified, more productive regrowth forests substantially different in structure and composition. Now it is recognised that the maintenance of biodiversity associated with structural and functional complexity of late forest development successional stages cannot rely solely on old-growth forests in reserves. Therefore, in managed forests, silvicultural systems able to develop or maintain old-growth forest attributes are being sought. The degree to which old-growth attributes are maintained or developed is called “old-growthness”. In this paper, we discuss silvicultural approaches that promote or maintain structural attributes of old-growth forests at the forest stand level in (a) current old-growth forests managed for timber production to retain structural elements, (b) current old-growth forests requiring regular, minor disturbances to maintain their structure, and (c) regrowth and secondary forests to restore old-growth structural attributes. While the functions of different elements of forest structure, such as coarse woody debris, large veteran trees, etc., have been described in principle, our knowledge about the quantity and distribution, in time and space, of these elements required to meet certain management objectives is rather limited for most ecosystems. The risks and operational constraints associated with managing for structural attributes create further complexity, which cannot be addressed adequately through the use of traditional silvicultural approaches. Silvicultural systems used in the retention and restoration of old-growthness can, and need, to employ a variety of approaches for managing spatial and temporal structural complexity. We present examples of silvicultural options that have been applied in creative experiments and forestry practice over the last two decades. However, these largely comprise only short-term responses, which are often accompanied by increased risks and disturbance. Much research and monitoring is required still to develop and optimise new silvicultural systems for old-growthness for a wide variety of forest ecosystem types.

1. Introduction

The global disappearance of primary, natural or unmanaged forests is of major concern (FAO, 2007). Many of these forests are old-growth forests, which provide numerous benefits and habitats unavailable in managed forests (e.g. Lindenmayer and McCarthy, 2002). The forests of Sweden and Finland provide examples of the effect of old-growth disappearance on various aspects of biodiversity. Many of these forests have been managed very intensively over the last 100 years. A comparison of abundance of various insects, birds, mammals, fungi, plants and lichen between intensively managed Swedish and Finnish forests and adjacent natural Russian forests revealed alarmingly a much lower number of species in the managed forests. These differences were attributed partially to the homogenized structure and reduced amounts of snags and woody debris in the even-aged monocultures (Berg et al., 1994; Angelstam, 1996). These, and other studies, suggest that the maintenance of key attributes of natural forests, as found in old-growth forests, is necessary to conserve a wide range of species.

Old-growth forests are a subset of primary forests that develop only under a limited set of circumstances, mostly associated with long periods without major natural disturbances. There are a number of approaches for defining old-growth forests (Wirth et al., 2009). One common approach, adopted in this paper, uses attributes of forest structure and composition, including a wide

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Table 1
Structural attributes commonly associated with different old-growth forests (examples from different forest types: Angers et al., 2005; Ansley and Battles, 1998; Dyne, 1991; Franklin et al., 2002; Franklin and Van Pelt, 2004; Holt et al., 1999; Kneeshaw and Gauthier, 2003; Meyer et al., 2003; Mosseler et al., 2003; Nilsson et al., 2002; Pollman, 2003; Salas et al., 2006; Siitonen et al., 2000; Tanouchi and Yamamoto, 1995; Trofymow et al., 2003; Tyrrell and Crow, 1994).

Old-growth structural attributes
High number/basal area of large trees
High stand volume or biomass
Large number/basal area of dead/dying standing trees
Large amount/mass of downed CWD
Wide decay class distribution of logs and/or snags
Several canopy layers/vertical variability
High number/cover of late successional/shade-tolerant species
High variation in tree sizes, presence of several cohorts
High spatial heterogeneity of tree distribution/irregular size and distribution of gaps
Thick forest floor
Special attributes (pit and mound relief, presence of epiphytes, presence of cavity-trees, tree hollows)
High variation in branch systems and crown structure/development of secondary crowns
Presence of advance regeneration

range of tree sizes and the presence of some old trees approaching their maximum longevity (Mosseler et al., 2003) (see also Table 1).

Approaches for maintaining old-growth attributes at stand and landscape scales include setting aside forests for preservation, in which no management takes place. Although highly desirable, in some regions ownership patterns or a high demand for wood products and other forest uses limits the application of this approach (Sarr and Puettmann, 2008). Furthermore, set-aside forests may be prone to natural disturbances (Spies et al., 2006). Areas outside reserves are also important, facilitating gene flow and migration of populations as well as providing complementary habitat (Lindenmayer and Franklin, 2002). It is therefore important to complement set-aside forests with managed forests that also reproduce key attributes of primary and old-growth forests, while, at the same time, addressing other social and economical management goals. This is particularly important in areas where reserves are too small to ensure the occurrence of natural disturbances within their boundaries or to accommodate all developmental stages of forest succession (Kneeshaw and Gauthier, 2003).

Although the primary old-growth forest area is still shrinking in many parts of the world, there are other areas, such as northeastern U.S., Japan and parts of central Europe, where the existing forests are ageing rapidly (Fig. 1) and thus offer new opportunities to increase the area of forest that can fulfil many of the functions and processes typically associated with old-growth (Davis, 1996).

Differences in ecological attributes between old-growth and forests managed for commodity products have been documented in a variety of settings (e.g. Perry and Amaranthus, 1997; Lindenmayer and McCarthy, 2002; Angers et al., 2005; Kenefic and Nyland, 2007). These differences need to be viewed in the context of temporal stand dynamics. Silvicultural practices focussed on wood production commonly result in production cycles of 25–150 years, whereas successional cycles of forests in some regions may continue over several hundred or a thousand years between stand-replacing disturbances (Scherzinger, 1996; Seymour and Hunter, 1999). As a result, managed forests often only cover 10–40% of the potential stand development period, and, consequently, many structural attributes of old forests (see Table 1) are absent or not fully developed in managed forests. In addition, forest harvesting and most other conventional silvicultural interventions do not aim to produce stand characteristics typically found in old-growth (e.g. Moore and Allen, 1999), but rather favour limited structures and tree species based on their economic value, rate of growth and management efficiency.

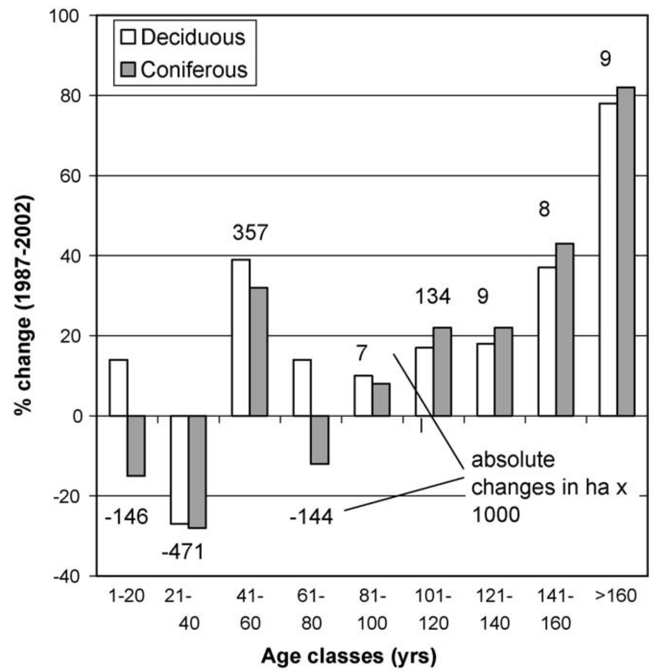

Fig. 1. The ageing of forests in Germany over the inventory period 1987–2002. Changes are depicted in percent and in absolute area (ha × 1000). Data only for former West-Germany (*Source*: National Forest Inventory).

It is now reasonably well understood that old-growth forests play an important role in harbouring of biodiversity (e.g. Lindenmayer and Franklin, 2002), in terrestrial carbon storage and sometimes sequestration (e.g. Carey et al., 2001) as well as in catchment hydrology (e.g. Vertessy et al., 1996) (see also Wirth et al., 2009). Concerns about their global disappearance have led to major efforts globally to increase the area of old-growth forests in reserves (e.g. USDA/USDI, 1994; DAFF, 2007). To establish such reserves, a definition of old-growth is needed that facilitates mapping and delineation of old-growth in the landscape. Yet the forests fitting one definition can vary widely in their ecological state, disturbance history and physical environments (e.g. Franklin and Spies, 1991). For this reason, the same authors introduced the term "old-growthness" to describe the degree to which forest stands express the various structural and functional attributes associated with old forests, and suggested that structural variability must be considered in our efforts to manage for old-growth (see also Fig. 2). Regrowth or secondary forests, relatively young forests that have regenerated after major disturbances, such as extensive cutting or wildfire (Helms, 1998), also can be highly variable with the same structural features found in old-growth to different degrees (Table 1). Evidence suggests that the occurrence of many, but not all, species typically found in old-growth is linked to specific structural attributes and not to old-growth as such (e.g. Siitonen and Martikainen, 1994; Gibbons and Lindenmayer, 1996; Lonsdale et al., 2008). Thus the strict separation of forested landscapes into old-growth and regrowth forests (Fig. 2) may not represent an optimal species conservation strategy with regard to the provision of habitats in the landscape. Instead, it may be better to manage forests for conservation based on their degree of old-growthness, their local and landscape functions in recognition of the expected opportunities for, and constraints to obtaining desirable levels of old-growthness. However, practically, it could be extremely difficult and costly to evaluate and assign a specific

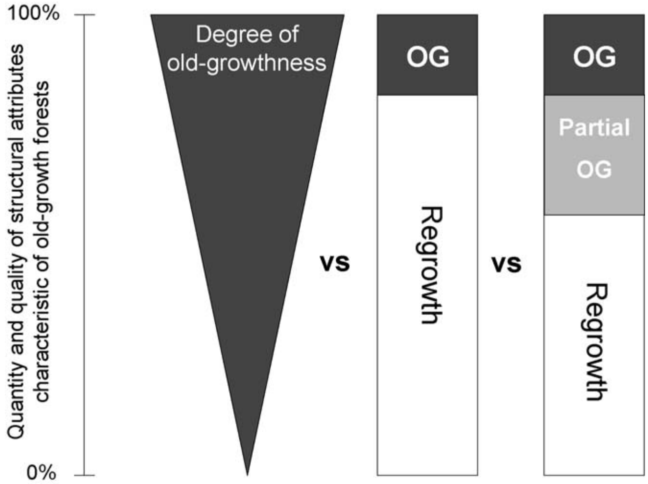

Fig. 2. Forests may be characterised according to their structural attributes (1) along a continuum of old-growthness, or (2) between old-growth and regrowth according to a certain threshold of old-growth attributes. The latter approach has the disadvantage of not distinguishing between regrowth forests with vastly different structural attributes that are reproducing to some extent old-growthness. A third approach might be the classification of three categories: old-growth, managed or regrowth forests with a substantial degree of old-growth attributes (partial old-growth), and intensively managed regrowth forests.

degree of old-growthness to each stand. Instead, a third category, partial old-growth or regrowth forests with some level of old-growthness, may be identified between true old-growth and intensively managed regrowth forests, a manageable approach to improve conservation planning (Fig. 2). The degree to which old-growth forests and old-growth structures should be maintained or restored at the landscape level is a complex, political question that requires an assessment of the trade-offs between different landscape values (i.e. Carey, 2003). This issue is outside the scope of this review, which focuses on management at the stand level. For the purpose of this review we have adopted a structure-based approach, and define old-growthness as a general aggregate measure of structural attributes listed in Table 1 for two reasons. Firstly, silvicultural practices modify stand structures and their dynamics directly, and secondly, information about links between stand structures, habitat provision and ecosystem functions is available (e. g. McElhinny et al., 2006).

This review emphasizes conditions for temperate and boreal forests because most studies investigating old-growth forests and their management have been conducted in these two biomes. Despite the large areas of old-growth forests found in the tropics, and their rapid disappearance rates, we have relatively little explicit information about them. Thus, while concepts discussed in this review also apply to tropical forests, specific examples are not provided.

2. Silvicultural approaches to maintain old-growthness

Three complementary approaches to the conservation and maintenance of old-growth forests and old-growthness have been termed reservation, retention, and restoration (Beese et al., 2003; Franklin et al., 1997; Keeton, 2006; Seymour and Hunter, 1999). The reservation of large patches of old-growth forests is an important element of an effective multi-scaled approach to the conservation of biodiversity at the landscape scale (Lindenmayer and Franklin, 2002). In this paper, however, we focus on the retention and restoration of structural attributes at the spatial scale of forest stands. Both are elements of a ‘‘coarse filter approach’’ to conservation, which aims to maintain biodiversity by providing a diversity of structures in stands as well as a diversity of ecosystems and their successional stages across the landscape (Noss, 1987; Hunter, 1991).

Silviculture is the manipulation of forest structures and dynamics to achieve management goals. Consequently, if reservation goals are met through passive management, as is often the case in existing old-growth forests, there is no need to implement silvicultural practices. However, in other settings, silvicultural practices may be beneficial or even necessary to promote old-growthness. These settings can be grouped into three categories:

(A) Current old-growth forests, resulting from the long-term absence of large-scale disturbances, and which are under consideration for management for timber production.
(B) Current old-growth forests, which are at risk of losing important elements of their structure or of being subject to intensive disturbances that they have not experienced historically. If, for whatever reasons, natural disturbances are unable to reduce this risk, active management may be required to maintain desirable attributes. We term this ‘‘cultural old-growth’’.
(C) Regrowth and secondary forests, which have been managed for other objectives, usually timber production, and are now targeted for the re-development of old-growth attributes.

In these three situations, silvicultural strategies aim at maintaining or increasing old-growth structural attributes in forest stands and hence also in the forested landscape (Fig. 3). Depending on existing forest conditions and economic, social, and political considerations, a combination of these strategies may be most suitable (Sarr and Puettmann, 2008).

3. Silviculture in old-growth forests available for timber production

When existing old-growth forests are to be managed for timber production, they will obviously lose their old-growth status according to most, if not all, definitions. However, to maintain a desirable degree of old-growthness in this situation, two options

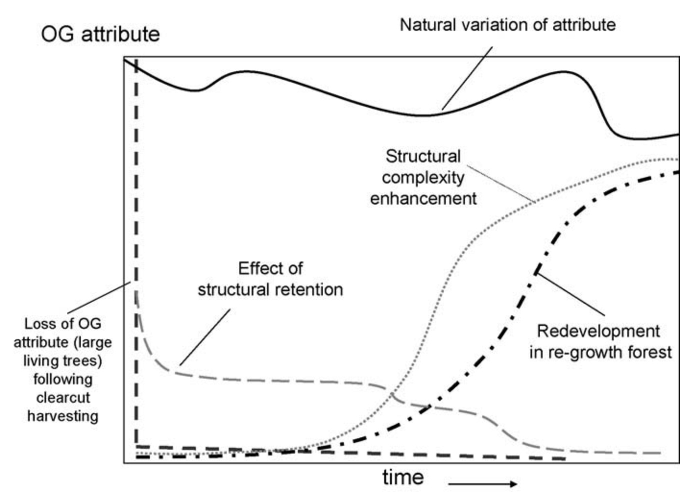

Fig. 3. Silvicultural strategies to maintain or increase old-growth structures in forest stands can rely on both retention as well as restoration to bridge or reduce the time in stand development in which structural complexity is low or certain structural elements may be missing, here for the example of large living trees. Continuous black line: temporal variation of structural attribute in natural old-growth forest. Dashed black line: loss of old-growth attribute following clearcut; dashed grey line: delayed loss of same old-growth attribute following retention of live trees. Dash-dot line: redevelopment of old-growth attribute in regrowth forest. Grey dotted line: accelerated redevelopment of same old-growth attribute in regrowth forest through restoration silviculture. Note different old-growth attributes might follow completely different patterns.

exist. In the first option, entire stands are managed using long production cycles, which extend well beyond the ages considered optimal for tree growth. Alternatively, selected trees, or other structural elements are retained in old-growth stands during silvicultural operations, while the rest is managed on shorter production cycles. This scenario is represented by the variable retention approach described by Franklin et al. (1997). We use the term production cycle instead of rotation, since the latter applies strictly to even-aged forests, while the former may be applied to individual trees and thus selection forests.

3.1. Maintenance of old-growth attributes through long production cycles

There are few incentives for managing forests on production cycles that are sufficiently long to include old-growth stages of stand development. For many tree species, the age at which the mean annual increment (MAI) culminates and subsequently declines is quite early relative to their potential maximum age. If landowners are more interested in maximizing the internal rate of return rather than stand growth, production cycles are short. Growing trees or stands to older ages may become economically attractive for species whose mean value increments culminate at an advanced aged. This happens when the decline in productivity after MAI culmination is slow and the market pays a significant size premium, and when the value of timber (on a volume basis) increases with increasing log dimension as often found for high quality hardwoods. On the other hand, recent advances in sawmill and lamination technology have more or less eliminated the size premium for standard quality conifer timber, providing little incentive to produce larger softwoods over longer production cycles. In some settings, landowners may even obtain a lower price for larger logs (e.g. Eschmann et al., 2003). The higher risk of disturbance in long production cycles introduces another concern. For example, as trees age and become taller they become more susceptible to windthrow (Peltola, 2006), more easily water-stressed due to the longer water transport distances between fine-roots and the crown (Ryan and Yoder, 1997), and hence are more susceptible to secondary pathogens such as bark beetles that affect stressed or weakened trees (e.g. Kelsey and Joseph, 2001). Given the uncertainties of future climatic conditions, risk-adverse forest managers will likely shorten production cycles (Kellomäki et al., 2000). An additional factor favouring short rotations is the risk to wood quality, such as discoloration or fungal colonization of stem wood, which increases with age in many tree species (e.g. Knoke, 2003). Therefore extended production cycles are unlikely to be adopted in production forests in the absence of some type of financial compensation from government or private conservation organisations (e.g. Küpker et al., 2005).

However, extended production cycles can have some financial advantages and environmental benefits such as reduced costs for regeneration-related management activities, higher diversity of products or wildlife habitat, hydrological benefits, and increased carbon storage (Curtis, 1997). Also, potential damage from fires may decline with tree age; as tree crowns rise, inter-tree spacing usually increases, and the bark becomes thicker, providing better insulation against damaging temperatures (e.g. Wyant et al., 1986). In some forests, however, the opposite may be the case when fuel ladders develop with age (Spies et al., 2006).

Examples of species and settings that resulted in long production cycles in managed forests include oaks (*Quercus*) grown for veneer in central Europe, where production cycles may extend to 200–300+ years (Vanselow, 1960). Harvey et al. (2002) present a case for cohort-based stand management with the goal to ensure a proportion of late-successional stands in the southern-boreal forest landscape of Québec. Here, long rotations do not equate to long production cycles for one species, but to the successional pattern of cohort replacement from early to mid and late successional stands that differ in species composition. Through uneven-aged silviculture, the advanced regeneration of shade-tolerant species can be recruited for successively older stands. This cohort-based approach may be more widely applicable in forests undergoing stand-replacing disturbances and distinct successional species replacement.

It is important to realise that extended production cycles, by themselves, can make only a small contribution to increasing the degree of old-growthness. Only attributes linked to large tree dimension and associated spatial patterning automatically benefit from implementation of long production cycles. The majority of attributes listed in Table 1 require additional management efforts, such as specific retention or restoration prescriptions.

3.2. Retention of old-growth structures

Many foresters recognise the benefits of regeneration methods modelled on natural disturbance dynamics to meet the establishment and early growth requirements of desired tree species. In forests subject to periodic stand-replacing disturbances, and where the target tree species have pioneer characteristics, clearfelling systems have been adopted (e.g. Hickey and Wilkinson, 1999; Bergeron et al., 2001). The application of clearfelling has been very successful in the regeneration of selected tree species. However, over time it has been recognised that forest structure and associated functions and processes differ in many ways between naturally disturbed and clearfelled forests (Lindenmayer and McCarthy, 2002; Pedlar et al., 2002). Even intensive natural disturbances leave behind dead or living structural elements, termed "legacies" (Franklin et al., 1985). The role of these legacies, or residual structures, for conservation of biological diversity and the recovery of ecosystem functioning following disturbance is well recognized (Jonsson et al., 2005; Vanha-Majamaa et al., 2007). Many studies have documented the relationships between the occurrence and abundance of such structural attributes and the occurrence, abundance and diversity of different taxonomic groups (Lindenmayer and Franklin, 2002). Differences in stand structures following natural or silvicultural disturbances have been documented for both stand-replacing disturbances and small scale, gap-phased disturbances (e.g. Coates and Burton, 1997; Spies and Franklin, 1989). For example, the implementation of uneven-aged selection systems has led to a substantial lack of old-growth attributes in a variety of ecosystems (Kenefic and Nyland, 2007; Angers et al., 2005; Müller et al., 2007). Furthermore, the size distribution and spatial arrangement of gaps is more uniform in selectively logged stands (Puettmann et al., 2008). However, simple changes in management practices may counteract these trends. For example, reducing the degree of tree utilisation can increase the abundance of CWD in selection or other silvicultural systems to levels higher than in unmanaged forests. While these inputs are often only temporary, in forests with CWD decomposition times substantially exceeding the interval between harvests, such periodic inputs could sustain abundant downed wood continuously (e.g. Doyon et al., 2005; Goodburn and Lorimer, 1998).

The structural simplification of selection forests demonstrates that retention of structural attributes should be considered in treatment prescriptions for uneven-aged silvicultural systems where appropriate. However, in this context the term "retention" implies that an attribute that would be removed under conventional management is deliberately retained for conservation purposes. This is fitting for most structural attributes retained in a modified clearfelling system, such as variable retention harvesting. The term is less appropriate for selection systems. In these systems, standard operations leave much of the stand behind. In these settings the term retention should be limited to structural attributes, such as dead trees, habitat trees, or non-vigorous and low quality trees, which would be removed under conventional uneven-aged management. Modified prescriptions to maintain undisturbed stand patches or an intact understorey may be more appropriately called "restricted selection". In addition to retained structural attributes, the spatial and size distribution of gaps is important for emulating patterns created by natural disturbances (e.g. Coates and Burton, 1997).

The retention of structural elements at the time of harvesting is based on two assumptions: 1) retained structures help maintain a higher level of biodiversity and ecosystem functioning on site than that attained without them, at least in the short term; and 2) retained structures facilitate the rapid recovery of biodiversity and ecosystem functioning (also called "life-boating" hypothesis).

The structural elements that can be retained (Tables 1 and 2) range in spatial scale from individual trees to large patches of vegetation. The list of structural attributes characterising old-growth conditions is remarkably similar across many different forest types. However, owing to the lack of a common inventory protocol, these attributes have been quantified differently in each study, making it extremely difficult to aggregate the information.

The benefits of retaining selected structural attributes have been demonstrated in numerous studies, especially in the context of the first assumption. Recent reviews (e.g. Lindenmayer and Franklin, 2002; Rosenvald and Lõhmus, 2008) concluded that these two assumptions are met in most situations. However, a detailed review of the large body of literature on effects of retention is beyond the scope of this paper. The following studies provide examples of the benefits of retention on birds and understorey (Merrill et al., 1998; Beese and Bryant, 1999), canopy lichens (Coxson and Stevenson, 2005), aerial insects (Deans et al., 2004), ground-layer bryophytes (Dovčiak et al., 2006), small terrestrial mammals (Gitzen et al., 2007) and saproxylic beetles (Jonsell and Weslien, 2003). Since the retention of structural attributes has not been practised for very long, short-term responses (assumption 1) are documented better than the long-term responses (assumption 2). In relation to the second assumption, short-term studies can only document the presence of propagules and sexually mature organisms in retained structures, and few studies have investigated the recolonisation of harvested areas originating from retained vegetation patches (e.g. Fisher and Bradbury, 2006; Tabor et al., 2007).

For management purposes, it is particularly important to learn how ecosystems respond to varying degrees of structural attributes retained and to different spatial arrangements of these attributes (Table 2). This scientific information is available only for few ecosystems and, even then, only to a limited extent (Lindenmayer and Franklin, 2002). The ecological effects and

Table 2
Silvicultural considerations regarding the retention of structural attributes.

Structural attribute to be retained	Desired density	Preferred spatial arrangement	Stability and dynamics	Risks and undesirable effects	Other management considerations
Live and habitat trees	Depends on habitat requirements of species that use trees and on spatial extent of desirable (e.g. seed dispersal) and undesirable (regrowth suppression) effects	Dispersed to meet ecosystem functions over entire area (seeds, water table, habitat)	Depends on windthrow risk (exposure, tree parameters, soil type)	Exposed and stressed trees may become breeding sites for secondary pathogens such as bark or jewel beetles	Retention of individual trees more hazardous than aggregates during harvesting and site preparation
		Aggregated to maintain forest conditions in patches, to reduce windthrow, to facilitate slash burning, and to reduce overstorey competition	Uprooted or snapped retention trees serve as dead wood	Suppressive overstorey effects on growth of regeneration and regrowth	
Standing dead trees	Depends on habitat requirements of species that use them, but see risks	Dispersed to serve as habitat for saproxylic organisms	Low windthrow risk, high risk of burning. Durability depends on decay resistance	Safety concern near roads, tracks and other frequented places	
		Aggregated to ensure safety of forest workers and visitors			
CWD on the ground	Same as above	Dispersed distribution preferred for many organisms with very low mobility and small home ranges	Persistence depends on decay resistance of species and log dimensions. Recruitment from snags and live trees required to maintain pool	Fresh logs as breeding ground for pathogens (e.g. bark beetles)	Large CWD as obstacle for machine based operations. Placement should take extraction system into account
		Aggregation reduces obstacles for future forestry operations		Might increase severity of fires, also problem of smoldering	
Patches with undisturbed vegetation incl. advance regeneration	Depends on size and functions. As source of propagules, dispersal distances should be considered. Edge effects into patches should be minimised	Dispersed retention not possible	Stability depends on size, edge effects, and exposure to wind and fire. Some disturbance within patches is not incompatible with retention goals	Same risks as for above attributes. Patches may harbour browsing animals that affect regeneration	Large patches are operationally easier than many small ones, in particular, where fire is required for site preparation

The desired attributes, their spatial arrangement, stability and associated risks will depend on forest type, disturbance regime and retention objectives. The desired density of structural attributes also always depends on the level of acceptable production losses.

tradeoffs of varying degrees of retained structures are being assessed in several experiments, for example, by the Ecosystem Management Emulating Natural Disturbances (EMEND) project in Alberta (Spence and Volney, 1999). Results from EMEND confirm that different minimum retention levels are needed for different organisms (Gandhi et al., 2004).

In the evaluation of spatial patterning of retention structures, the comparison of dispersed versus aggregated retention has received some attention (Franklin et al., 1997, 1999). A listing of advantages and disadvantages of these two approaches in relation to various retention goals shows that neither approach provides a consistently better achievement (Franklin et al., 1997). Consequently, these authors generally recommend a combination of dispersed and aggregated retention ("variable retention").

Silvicultural prescriptions are specific solutions to a specific set of circumstances and management objectives (Smith, 1962). Thus, retention prescriptions that become part of a silvicultural system need to fit in with this constraint. Important considerations for incorporating the retention of structural attributes into silvicultural systems include the desired density and distribution of retained structures, their stability/longevity and how the spatial arrangement may influence the risks associated with specific retention practices and the effects on forestry operations (Table 2). The desired density of retained structural attributes, as listed in Table 2, depends, to a large degree, on the specific species, species groups or the processes of concern. For example, to maintain saproxylic insects, the density and distribution of CWD of preferred and required wood characteristics is important (e.g. hardwood vs. softwood, or specific species). In addition, the proportion of CWD in the various decay classes, and the range and distribution of log dimensions may be important for providing continuity of habitat in space and time for species with different mobility (Grove, 2002). Thus, in the absence of better information, varying retention densities in both space and time may be the best approach for maintaining processes, and providing habitats for a wide range of organisms.

Although a topic of much study, we do not yet understand fully how various organisms or ecosystem functions respond to various amounts and arrangements of CWD (Harmon, 2002). However, even if good information on ecological responses were available, the question of what constitutes satisfactory amounts and distributions of retention attributes is still largely subjective, based on factors such as acceptable economic impact, conservation status of species affected, and associated risks.

Similarly, as a result of the fairly recent interest in variable retention, knowledge about the stability and long-term dynamics of retained structural elements is limited. The susceptibility of standing structural attributes to uprooting, snapping or otherwise succumbing to the influences of wind, fire, dieback and pathogens is, in many situations, likely to be higher than in an intact forest (Bladon et al., 2008). Information about the post-harvest dynamics of these structures is important if their functions are to be maintained over a full production cycle (Table 2). For example, retained live trees have some functions that are important only for the initial recolonisation phase, such as soil protection, provision of seeds or serving as an inoculum for mycorrhizal fungi (e.g. Outerbridge and Trofymow, 2004; Rosenvald and Lõhmus, 2008). However, in the long-term, they generate large trees and crowns, snags, and downed wood.

Typically, structural attributes in old-forests develop under conditions that do not prepare these attributes for sudden exposure in post-harvest situations. Therefore wind damage of retained trees and vegetation patches is a common phenomenon in retention harvests, especially shortly after harvesting (e.g. Coates, 1997; Scott and Mitchell, 2005). These concerns can be offset partially by carefully planning the location and orientation of retention patterns. For example, wind damage (uprooting and snapping) increases with decreasing density of retained trees and is more pronounced for trees in dispersed than in aggregated retention patterns (Esseen, 1994; Moore et al., 2003). In addition, trees with low height-to-diameter ratios, sparse crowns, greater crown length, and those belonging to deep-rooting species are less susceptible to wind damage (Moore et al., 2003; Scott and Mitchell, 2005) and should be selected preferentially in areas where wind damage is of concern.

Alternatively, if fire, is the major disturbance agent, other aspects are of concern to ensure long-term benefits of the retained structures. For example, slash loads around retained trees may need to be reduced to ensure tree survival in the event of fire (e.g. Neyland, 2004). The need for such treatments can be minimized by adopting aggregated retention patterns in the interior of cut blocks. Sudden openings in the canopy layer may not lead necessarily to instant mortality, but can lead to increased physiological stress in retained trees. Trees of different species or sizes may be affected by stress to different degrees (Laurance et al., 2006). For example, dominant trees with large crowns may be more susceptible (Laurance et al., 2000) to stress owing to increased water demands (Bladon et al., 2005, 2007). In these instances, tradeoffs between wind-firmness and tolerance to water stress factor in decisions about which trees to retain.

After an initial period of instability after harvesting, in which the least stable and least resistant individuals tend to die, mortality of retained trees is likely to decline over time (e.g. Bebber et al., 2005). Tree mortality may not be undesirable when linked to certain structural objectives. It serves as input into the CWD pool, and can lead to other important microhabitat features, such as pit-and-mound topography resulting from windthrow (Bauhus, in press). Thus, typical attrition rates of retained trees and vegetation patches need to be factored into designs of retention levels and spatial patterns (Vanha-Majamaa and Jalonen, 2001; Cissel et al., 2006).

Retained structural elements also can have undesirable effects and pose risks (Table 2). Growth reduction of new tree cohorts caused by competition from the retained overstorey trees, and the risk of the spread of pests and diseases propagating in retained structures are the main concerns. Such growth reductions have been documented in many studies (e.g. Bauhus et al., 2000; Bassett and White, 2001; Rose and Muir, 1997). However, the magnitude of growth reduction appears to be highly variable and depends on a range of factors such as size and vigour of retained trees, shade tolerance of the establishing understorey, site resource availability, and spatial patterns of retained trees. Reductions appear larger on sites with low productivity, probably due to the combined effects of shading and root competition. Where retained trees suppress vegetation outside their crown projection area (Puettmann and D'Amato, 2002), competitive effects of overstorey trees in aggregated retention most likely will be less than in dispersed retention, particularly when shade-intolerant species dominate the recruitment layer (e.g. Palik et al., 1997).

Retained trees that become stressed due to sudden exposure after harvesting are likely to be more susceptible to secondary pathogens. Furthermore, some damaging insects may benefit from the warmer microclimate after harvesting and the provision of fresh breeding and foraging material in abundant CWD. This may be particularly problematic for coniferous forests, where bark beetles are important pest species. Factors, such as whether insect populations are at endemic levels or how many damaged trees are available for insects, can determine the size of bark beetle or pine shoot borer (*Tornicus* sp.) populations in a restoration area with retained trees and CWD (Eriksson et al., 2006; Martikainen et al., 2006). Under favourable conditions, the damage from insects and diseases after retention harvests does not necessarily exceed that under conventional silvicultural systems. However, in regions with warmer climates than in the boreal forest example above, insect populations may become more responsive and be of greater concern. It may therefore be advisable to retain trees and CWD of species that are less susceptible to insect damage (Vanha-Majamaa and Jalonen, 2001), or to use fire to lower the suitability of CWD as breeding material (Eriksson et al., 2006).

The density and spatial arrangement of retained structures have a variety of other implications for forestry operations and ecosystem development (Table 2; see also Franklin et al., 1997; Beese et al., 2003). Given the complexity of factors and their potential interactions, it is not surprising that many large-scale, operational-size experiments are currently being conducted to investigate the effects of different retention strategies on ecological, economic and social forest values (e.g. Coates et al., 1997; Abbott et al., 1999; Spence and Volney, 1999; Brown et al., 2001; Brais et al., 2004; Poage and Anderson, 2007). The degree to which results from these experiments are specific to their local forest types or the extent to which they can be extrapolated, is a question of great importance, since these research efforts are concentrated in temperate and boreal regions and similar studies are lacking in the tropical and subtropical forests.

4. "Cultural old-growth" forests

In several definitions, old-growth forests have been characterised by a long-term absence of intensive disturbance. However, in some old-growth forests, regular minor disturbances are required to maintain old-growthness or to stabilise forest structure (Kaufmann et al., 2007). Well-known examples of this type of forest include the ponderosa pine forests in western North America (see Kaufmann et al., 2007 for more examples). While stand-replacing fires are rare, these forests were subject to frequent (3–38 years) low intensity fires in the pre-European era. Native Americans likely had a major influence on this fire regime to encourage development and fruiting of plants, to increase the abundance of selected species while discouraging others, and to facilitate hunting (Hessburg and Agee, 2003). Other opinions suggest that Native American burns only supplemented or substituted for natural lightning fires in these fire-prone environments (Baker, 2002). However, the fire-regimes changed substantially in many places with the landscape changes subsequent to the arrival of European settlers (Hessburg and Agee, 2003). Through a reduction in fire-frequency, mainly due to grazing and fire suppression, a number of ecosystem characteristics changed. Specifically, forest floor depth and fuel loads increased, as did tree densities, particularly of shade-tolerant and fire sensitive conifers such as Douglas fir and true firs (e.g. Covington et al., 1997). These changes led to reduced soil moisture and understorey vegetation diversity, and to increased mortality of old trees (Binkley et al., 2007). As a result of the increased amounts of fuel, continuous canopy and fuel ladders, high intensity crown fires are likely to be stand-replacing events. Restoration efforts, which include the removal of trees and ground fuel through thinning and controlled burning, maintain open stand structures that prevent or reduce the likelihood of high-intensity fires (Covington et al., 1997). Where the maintenance of old-growth structures is dependent on active management of disturbance regimes, as in the example above, we might speak of "cultural old-growth". In addition to the maintenance of disturbance regimes that have shaped these forests, additional restoration management may be necessary, as outlined below.

5. Restoring old-growth attributes in regrowth and secondary forests

Much of the forested area previously covered by old-growth in temperate, Mediterranean and subtropical regions has been converted to regrowth or secondary managed forests with substantially different structures and, in many cases, different species compositions as well (Sands, 2005).

A change in management objectives towards encouraging development of old-growth structures requires a shift in management approaches and practices. In stands where management was highly intensive and successful at homogenizing composition and structure, this shift requires a longer time period for successful "transformation" or "conversion" (Kenk and Guehne, 2001; Kuuluvainen et al., 2002). In many parts of Finland, for example, aspen has been removed almost completely, and considerable effort is required to bring large aspen trees back (Vanha-Majamaa et al., 2007). However, stands in which management was not aimed at, or was unsuccessful at homogenizing composition and structures may have many of the desired structural attributes already, and thus require less restoration effort (Newton and Cole, 1987).

From a landscape perspective, restoration can be used to complement conservation efforts (1) in reserves to enhance habitat quality and quantity, (2) in multiple-use forests between small and fragmented reserves to complement habitat and improve connectivity, and (3) to create buffer zones between reserved and intensively managed forest areas (Kuuluvainen et al., 2002).

Restoration practices mainly aim to increase structural complexity of forest stands (see McElhinny et al., 2005, for a definition of structural complexity). This may be achieved through the management of density and tree regeneration (Kenk and Guehne, 2001; O'Hara, 2002; Choi et al., 2007; Davis et al., 2007). While these two aspects are part of "traditional" silvicultural practices, the new suite of restoration objectives provides unique challenges. For example, while traditional silvicultural systems are designed to optimize conditions for regenerating seedlings, overstorey densities specified in restoration treatments may be driven by wildlife habitat objectives, which are suboptimal for regeneration (Puettmann and Ammer, 2007). Furthermore, silvicultural restoration prescriptions need to address a variety of other components of stand structure and composition, such as canopy and crown structures as well as understorey vegetation typically found in old-growth (Table 1) (Franklin et al., 1981; Davis, 1996).

The list of structural components found in old-growth forests (Table 1) does not provide information about their relative importance, which is likely to vary among different stand types, ownerships and regions (Mansourian et al., 2005). Developing a hierarchy of priorities for the desired structural and composition components (Table 1) will help to resolve potential conflicts. In regions in which present old-growth can be used as a blueprint for management efforts, structure and composition targets can be quantified in detail (e.g. Cissel et al., 2006; Bergeron et al., 2001). In areas where old-growth is absent or limited, desired future conditions may need to be more generic (Zerbe, 2002; Mansourian et al., 2005). Specific structure and composition goals can be derived either from historical evidence, or an understanding of habitat requirement of selected species or taxonomic groups (e.g., Conner and Rudolph, 1991; Thompson et al., 2003). The latter can be regarded a fine-filter approach to conservation (Hunter, 1991) in contrast to broader goals of management for structural complexity.

A specific list of attributes considered essential or desirable goals for management (Table 1) together with an inventory of current conditions provides an information base for assessing which strategies are best suited to achieve the goals (e.g. Schmoldt et al., 2001). Besides ecological constraints, concerns about costs, social acceptability and short-term negative impacts of necessary practices are important, and may influence the decision whether to use a passive, reserve-based approach towards increasing old-growth, or an active management approach. Kuuluvainen et al. (2002) provide some good examples of active management approaches for increasing old-growth attributes relatively quickly. The potential benefits of an active management approach rely on two basic assumptions (see also Keeton, 2006):

(1) Active management can accelerate the development of old-growth structural attributes in forest stands (Fig. 3).
(2) Active restoration of old-growth structures offers additional advantages over passive (non-manipulative/unguided) restoration, including higher predictability and reduced risks, and a higher level of provision of goods and services, such as timber.

Initial approaches to restoration suggested adopting "traditional" uneven-aged silvicultural practices (Benecke, 1996; Emmingham, 1998). However, structural goals and associated constraints and conditions in managing for old-growthness are quite different from the conditions that have led to the development of current silvicultural systems (Puettmann et al., 2008; for examples about the impact of such differences see Kenefic and Nyland, 2007). Traditional silvicultural systems were developed for efficient timber production in intensively managed, homogenized forests (Puettmann et al., 2008). In contrast, restoration goals for old-growthness typically focus on increasing structural complexity (Keeton, 2006). Because of the limited scientific information currently available to guide our efforts, various research programs were initiated in the 1990s to investigate whether management could accelerate the development of old-growth structural components, and also the potential benefits of an active restoration approach (e.g. Poage and Anderson, 2007; Seymour et al., 2006; Kuehne and Puettmann, 2006). Most of these studies are relatively recent and information about many aspects, especially long-term responses, is still rare. The following section reviews our current understanding of the two above-mentioned assumptions for a variety of stand components.

If a few, large trees are a desirable characteristic of future stands, the average tree response, which is often documented in thinning studies, is not a useful measure. The largest trees in a stand appear to be influenced less by the overall competitive conditions in the stand or by their local neighbourhood (D'Amato and Puettmann, 2004; Simonin et al., 2006). Consequently, thinning intensities around these trees need to be higher than in "standard" thinning prescriptions to achieve a substantial growth response (Davis et al., 2007).

Criteria for tree retention need to acknowledge desirable future species compositions and structure. Typically, managed stands comprise a limited set of crop tree species. However, even managed plantations often contain a few trees of non-crop species, which usually have regenerated naturally (e.g. Keenan et al., 1997; Davis et al., 2007). These trees may have little economic value and therefore are discriminated against in release or thinning treatments as potential competitors (Walstad and Kuch, 1987; Mason and Milne, 1999). However, they become of greater interest as residual trees in restoration treatments to increase the diversity of species and structural conditions in the stand. These less desirable tree species, if left during thinning operations can make a significant contribution to the seedling bank and thus on future development of a stand towards the composition of old forests (Keeton and Franklin, 2005; Kuehne and Puettmann, 2008). Practices required to ensure the survival of ecologically important midstorey species could include removal of overtopping trees, even potential crop trees (e.g. Welden et al., 1991).

Similarly, selection of cut-and-leave trees may be altered to provide for a variety of crown structures. For example, forked trees, or trees with cavities or diseased or damaged tops may provide unique habitat features, but typically are marked for removal because of their low value (Kenefic and Nyland, 2007). Another argument for their retention is that the economic benefit of selling such (non-crop) trees is often relatively small. Restoration activities also may aim at actively preventing the mortality of cavity trees during management activities (Conner et al., 1991; Bull et al., 2004; Kenefic and Nyland, 2007). Mortality of cavity trees is typically higher than that of healthy trees (Conner et al., 1991), and decisions about thinning densities should consider leaving extra trees, which are designated as potential future cavity trees (e.g. Cissel et al., 2006).

While it has been shown that the development of many tree attributes can be accelerated through management activities (e.g. Choi et al., 2007), information about the influence of restoration activities on other attributes is lacking. For example, development of certain crown structures, such as dead branches, has been documented in old forests, but not in response to thinning in mature or old forests (Ishii and McDowell, 2002; Ishii and Kadotani, 2006). The experiences from thinning studies in young stands, when crowns consist largely of small or semi-permanent branches, may not be transferable.

In a variety of ecosystems, the species diversity and biomass of understorey vegetation has been shown to increase after thinning or partial cuts (West and Osler, 1995; Bailey and Tappeiner, 1998; Bauhus et al., 2001). Several studies showed that the degree of thinning related positively to the increase in understorey vegetation diversity and biomass (Harrington and Edwards, 1999; Battles et al., 2001; Elliot and Knoepp, 2005). The initial response of understorey vegetation appears to be a combination of a response to the harvesting disturbance and changes in the availability of resources such as light and water. Moreover, the interplay between these factors may lead to a decline in understorey cover (Thomas et al., 1999; Davis and Puettmann, in press). For example, shrubs injured in logging operations are unable to exploit increased resource levels until they recover from damage (Kraft et al., 2004; Davis and Puettmann, in press). Unfavourable microclimatic conditions, such as lower humidity, are probably also responsible for the initial decline of mosses after thinning (Davis and Puettmann, in press). On the other hand, herbaceous species increase in diversity and abundance quickly, but over time will be repressed by regrowth of overstorey trees and shrubs (Beaudet et al., 2004; Davis and Puettmann, in press).

Initially thinning appears to alter species composition towards early successional species (Griffis et al., 2001), a trend contrary to that found in unmanaged old-growth forests (Keenan et al., 1997; Schoonmaker and McKee, 1988). However, after longer periods without larger disturbances, understorey species composition in thinned stands becomes more similar to old-growth than in unthinned stands (Bailey and Tappeiner, 1998; Lindh and Muir, 2004). This is probably due to the recovery of the overstorey cover after thinning (Davis et al., 2007; Maas-Hebner et al., 2005), which has been shown to reach overstorey cover levels (He and Barclay, 2000) and leaf areas (Bailey and Tappeiner, 1998) similar to unthinned and old-growth stands within two to three decades. The resulting reduction in light levels (Beaudet et al., 2004) and below-ground resources (Riegel et al., 1995) in conjunction with plant interactions among understorey plants, such as competitive and facilitative processes (Thomas et al., 1999; Delagrange et al., 2006) are most likely responsible for the shift in species composition.

Restoration efforts to influence the understorey, in many cases will influence tree regeneration as well. For example, advanced regeneration is important for future dynamics of forest ecosystems as it facilitates an increase in species diversity and hence quality of different canopy layers (Mesquita, 2000; Murphy et al., 1999). The establishment of a vigorous tree understorey provides an important functional component for resiliency and adaptability of such ecosystems as advanced regeneration can usually respond quickly to overstorey mortality or removal.

Fully stocked, homogenous stands can be manipulated easily to increase structural and environmental variability within a stand. Even in the absence of specific planning, inherent stand variability and logistic constraints are likely to create some spatial heterogeneity following thinning (Berger et al., 2004). In restoration treatments, structural variability can be generated if criteria other than spacing are used in prescriptions. For example, management based on tree size, e.g. diameter-limit cuts or target diameter harvesting, leads to increasing small-scale spatial variability (Angers et al., 2005). Prescriptions also can include a wide range of residual tree-to-tree distances or gaps, and leave unthinned islands (Cissel et al., 2006). The latter may be regarded as aggregated retention in the thinning phase. However, gaps, small openings or evenly spaced canopies may close relatively quickly by lateral branch expansion and vertical growth of mid and understorey trees such that these openings are only a temporary feature (van der Meer and Bongers, 1996; Splechtna et al., 2005).

Most managed forests contain lower CWD levels than old, unmanaged forests (e.g. Morgantini and Kansas, 2003; Ekbohm et al., 2006). The recognition of the importance of CWD has led to a range of active and passive approaches to increase the woody detritus pool in managed forests (see Table 3), although, in most cases, it is very difficult to determine the quantity and distribution of CWD required to achieve certain management objectives (Harmon, 2002). Active approaches comprise girdling or poisoning to create standing dead trees, and felling and pulling to create CWD on the ground (e.g. Keeton, 2006). In addition, leaving more slash, including trees, after harvesting as well as burning to kill some live trees are means to increase CWD at the time of harvesting (e.g. Vanha-Majamaa et al., 2007). However, in continuous-cover forestry and restoration practice, active creation of CWD is likely to be restricted to special situations, for example where there is an immediate need to provide habitat for threatened organisms (e.g. Filip et al., 2004) or where, in the absence of woodpeckers, such as in Australia, cavities take a very long time to develop (Gibbons and Lindenmayer, 2002).

Table 3
Structural attributes of old-growth forests and silvicultural approaches to promote these (expanded from Keeton, 2006).

Desired attribute	Silvicultural interventions
Vertical canopy stratification	• Selection cutting • Continuous regeneration and its release
Horizontal variation in stand density	• Group selection and gap harvesting • Variable density thinning
Presence of large trees	• Crown thinning to release and increase growth of most vigorous trees • Long rotations
Presence of standing dead trees	• Allow self-thinning • Tree girdling or poisoning • Burning • Permanent retention of live trees • No or limited salvage following disturbance
High levels of fallen CWD	• Allow self-thinning • Tree felling or pulling • Permanent retention of live trees • No or limited salvage following disturbance • Lower utilization standards and leave more slash
Dead wood in crowns	• Long rotations • Manipulation of crown expansion and retraction
Presence of late successional mid and understorey vegetation	• Maintain unthinned stand areas

Passive approaches to increase CWD can rely on density-dependent (competition driven) and density-independent mortality. Density-dependent mortality as a result of self-thinning is particularly high in young even-aged stands or groups. Ferguson and Archibald (2002) showed that the basal area of dead standing trees was closely related to the amount of live tree basal area in fire-origin boreal forests of northwestern Ontario. Thus, what might be a suitable practice to promote late-successional understorey (see above) is also suitable for the passive creation of dead wood (e.g. Vanderwel et al., 2006). However, a large proportion of this material may be small in size, and therefore unsuitable for particular types of saproxylic organisms.

Density-independent mortality, which may be between 1 and 2% per annum in mature and old stands (Van Mantgem and Stephenson, 2007; Lewis et al., 2004) ensures a constant supply of dead wood. If individual or groups of dying or dead trees are not salvaged, even after disturbances, or salvaging is reduced, the input of CWD could be increased considerably. Bouget and Duelli (2004) argue that, even in coniferous forests with the risk of bark beetle infestation, windthrow gaps can be managed in an adaptive way that allows the retention of freshly created CWD islands.

By modeling CWD dynamics in Norway spruce stands, Ranius et al. (2003) demonstrated that the risk of losing sufficient quantities of CWD in the different decay classes is high, if insufficient live trees that can die over the course of a production cycle are retained. How much CWD persists over the course of a production cycle depends on the initial and continued input of dead trees and the decomposition rate of standing and downed CWD. To ensure a continual CWD input, it is important to retain live trees in a way that avoids high mortality rates soon after harvesting disturbance. Thus the maintenance of CWD is closely linked to the quantity and distribution of retained live trees (Table 2). The decomposition rate of CWD depends on a range of factors, including species-specific decay resistance, time until snag fall, stem size, climatic variables and the decomposer community (Mackensen et al., 2003; Ranius et al., 2003), all of which may need to be considered in an approach to maintain or increase dead wood. Lonsdale et al. (2008) have listed a number of examples where the application of best management practices has resulted in increased CWD levels. In addition, restoration practices aimed at creating CWD must be aware of possible conflicts with management of wildfire risk, insect pests and forest disease outbreaks. Lonsdale et al. (2008) discuss further issues related to dead wood management.

Just like any silvicultural treatment, constraints and risks of restoration treatments need to be evaluated carefully. Many restoration treatments are associated with substantial costs, which may prevent their widespread application, particularly on private land. Combining such treatments with harvesting operations that provide revenue and some form of compensation may be necessary for implementation (Keegan et al., 2002). Furthermore, restoration treatments may lead to increased risk of disturbance, at least in the short term (e.g. Cremer et al., 1982). This is generally undesirable in forests also managed for timber production. For example, sudden canopy openings caused by intensive thinning or gap creation may lead to higher windthrow rates until trees stabilize through altered taper or crown dimensions (Mitchell, 2000; Achim et al., 2005). The potential for higher intensity fires may increase as understorey and midstorey vegetation layers and downed wood provide higher fuel loads (Agee, 1993).

To assess long-term development of stand structure and composition in response to alternative restoration options, increasingly silviculturists are using simulation models. Because of the higher predictability of tree development, most efforts focus on tree growth and mortality, with notable exceptions. Early attempts relied on standard growth and yield prediction models (e.g. Birch and Johnson, 1992). Alternatively, ecological gap models (Busing and Garman, 2002) or individual tree models provide more flexibility to simulate a variety of treatments (Choi et al., 2007). Most individual tree models have the limitation that they assume regular tree spacing. However, the recent development of spatially explicit models (e.g. SORTIE-ND: Coates et al., 2003), which may even include stochastic elements (e.g. LANDIS-II: Mladenoff, 2004), provide an opportunity to represent spatially variable treatments both within and among stands.

In summary, the development of many structural and composition components of old-growth stands can be accelerated through silvicultural interventions. However, the dynamics of the responses differ between ecosystems and initial conditions (e.g. Choi et al., 2007), and the timing and direction of the response of various structural components are not necessarily coupled. Some responses to restoration are very dynamic, e.g. increase in species diversity in understorey vegetation. Furthermore, structural components that are related to tree size can be manipulated efficiently through density management. However, secondary responses, e.g. wildlife populations or lichen communities, require much longer time periods to develop (Batty et al., 2003). The stand development stage, when the ecosystem is still or most responsive to restoration treatments, varies for the different structural attributes (Puettmann and Berger, 2006). To complicate things further, opposite response trends may occur. For example, advanced regeneration may develop into a dense midstorey layer that limits the development of the shrub and herb understorey. The complexity of interacting factors suggests that restoration should not be prescribed homogenously or at the stand level. Instead, decisions about priorities, timing, and what proportion of stands should provide what old-growth attributes of structure or composition may be necessary for efficient restoration efforts. Lastly, it is important to note that restoration treatments not only have to deal with logistical constraints and social acceptability, but they also need to deal with temporarily increased risks of disturbances.

6. Conclusions and outlook

Silviculture for old-growth attributes should not be considered as an oddity by foresters since the special ecological services that old-growth provides are becoming increasingly valued by society due to their rarity. Since silviculture is aimed at manipulating forest stands to achieve human objectives, managing for old-growthness is merely a new objective to add to the long list of the current ones. One of the main differences with previous objectives is that managing for old-growthness does not normally provide direct benefits to the landholder, but rather an indirect benefit to society as a whole. Consequently, in order to make silvicultural practices for old-growthness an attractive option, society as a whole would need to place a financial value on old-growthness. This is already occurring in some areas and countries, where government programs compensate private owners for foregoing harvesting, or for harvesting forests in unconventional ways. Similarly, certification could be considered as a kind of market incentive for maintaining old-growthness on some part of the managed landscape. While it may be feasible technically to retain and restore complex forest structures, silviculturists are also challenged to make these strategies work economically.

Here, we reviewed silvicultural approaches for old-growthness at the forest stand level. However, stand-level silvicultural strategies of course are influenced by the landscape or regional setting. Thus there are many other questions that need to be addressed at a larger scale to optimise silvicultural approaches. In this context we need to ask how much and where old-growthness should be maintained or developed preferably in the landscape, since the probability of disturbance changes with ecosystem type and landscape setting (Keeton and Franklin, 2004; Wirth et al., 2009). It will also be easier to implement complex structures in some parts of the landscape than in others (e.g. steep slopes).

An outstanding research question for managing for old-growthness concerns the quantity, spatial arrangement and temporal dynamics of forest structural attributes required to meet various management objectives.

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Disturbance, degradation, and recovery: forest dynamics and climate change mitigation

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Deforestation and sustainable forest management have been major international policy concerns since the 1970s. The potential implications of climate change as a result of increasing concentrations of atmospheric greenhouse gases has further focused policy attention on these issues. Greenhouse gas emissions associated with forest loss are currently estimated to contribute about 18-20% percent of global greenhouse gas emissions and an international financing mechanism to foster activities that reduce emissions from deforestation and degradation (REDD) has been put forward as part of negotiations to develop a new international emission reduction strategy following the first commitment period of the Kyoto Protocol.

Important methodological issues in accounting for reduced emissions from deforestation and forest degradation include establishing an appropriate baseline, ensuring additionality and preventing leakage. Other challenges include incorporating the effects of disturbances, such as fire or insect pests, successional dynamics and the longer-term impacts of historical and customary land use practices. Other management issues that need to be considered include the effect of changes in forest harvesting, silvicultural practices and biodiversity conservation objectives.

The paper proposes that, for greenhouse gas accounting purposes, forest degradation be considered as the long-term reduction in forest carbon stocks resulting from human-induced activities. Accounting for forest degradation needs to establish an appropriate baseline and provide for a balanced assessment of forest dynamics. Four activities that could potentially be considered forest degradation are presented: timber harvesting and wildfire in native forest in Australia and timber harvesting and shifting cultivation in PNG. The application of the definition of forest degradation to these indicates that if all associated carbon dynamics of these activities are considered, only selective harvesting in PNG results in a likely long-term reduction in carbon stock and therefore can be considered a form of 'forest degradation' for carbon accounting purposes. If the frequency of intense wildfires in Australia continues to increase and the link to human-induced climate change is clear, then this may also be considered degradation. The other two activities are examples of long-term cycles of felling and regeneration, where forest carbon stocks and productivity are being maintained.

Introduction

Deforestation is estimated to contribute about 18-20% of anthropogenic greenhouse gas (GHG) emissions (Baumert *et al*. 2005; IPCC 2007). Forests also remove carbon from the atmosphere due to increases in forest area and carbon uptake in regrowing forests and, on balance, there is estimated to be a net uptake of carbon in terrestrial ecosystems that varies considerably from year to year with varying climatic conditions and dynamics of large-scale disturbances such as wildfires (Raupach *et al*. 2007). A large proportion of these exchanges with the biosphere are not under human control (Kirschbaum and Cowie 2004). Emissions and removals from a limited set of activities (afforestation, reforestation and deforestation and, for some countries, forest, cropland and grazing land management) were included in accounting arrangements for

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Annex I parties in the Kyoto Protocol. Inclusion of reduced GHG emissions from deforestation and forest degradation (or REDD) in Non-Annex I countries have been actively promoted in the international negotiating agenda for the United Nations Framework Convention on Climate Change (UNFCCC) by the Coalition of Rainforests Nations (CRN) led by Brazil and Papua New Guinea. The Bali Action Plan, which launched negotiations for a new climate change agreement for the period post-2012, includes a direction for negotiating parties to address 'policy approaches and positive incentives' on REDD in developing countries. Support for including REDD in emissions reduction targets has been widespread but by no means universal (Fry 2008). Support is partly based on the argument that the cost of reducing emissions from this source is lower than the cost of other mitigating measures (Chomitz 2007; Stern 2007).

In order to claim credit for reducing GHG emissions from any source, it is first necessary to establish the baseline against which reductions can be assessed with some degree of reliability and objectivity (IPCC 2000; Brown *et al*. 2007). For the Kyoto Protocol, this was emissions in the calendar year 1990. In the case of REDD, this problem is compounded by a mixture of scientific and political arguments about what is considered to be 'forest' and what is meant by words like 'deforestation' and 'degradation' (Schoene *et al*. 2007). This paper focuses on the issue of forest degradation. The aim is to present a framework for analysing what constitutes 'forest degradation' using examples from Australia and Papua New Guinea and discuss some of the issues and challenges in measuring and accounting for forest degradation.

Defining 'forest degradation'

There currently no consistent, internationally agreed definition of forest degradation (Schoene *et al*. 2007). The UN Food and Agriculture Organisation (FAO) has been seeking to establish an international consensus on the definition of forest terminology since it began to collect information from member countries for periodic Global Forest Resource Assessments in 1947 (FAO 2003; Holmgren and Persson 2003). The UNFCCC also has processes and mechanisms for agreement to terminology and definitions, generally with the scientific support of the IPCC. Meanwhile, the perceived urgency of climate change as a policy issue has encouraged a variety of non-governmental organisations around the world to engage in their own assessments of the relationship between DFD and the global carbon cycle, using varying definitions and standards of measurement.

Forest degradation can be considered from a range of perspectives: forest productivity (products and services), genes, tree vigour and quality, species composition, soils, water, nutrients and the landscape (Schoene *et al*. 2007). It implies a long-term loss of capacity that may be difficult to assess, especially when applied to soils, water, and the landscape. Degradation from one perspective may also not mean degradation from all perspectives. For example, a change in species composition may not degrade a forest in terms of productivity or long-term carbon stocks but may result in a reduction in biodiversity or water yield (Penman *et al*. 2003).

The need to define forest degradation in the context of greenhouse gas inventories initially arose in response to concerns that selection of eligible activities under Article 3.4 of the Kyoto Protocol could give rise to an unbalanced accounting if certain types of activities were not included (Penman *et al*. 2003). These authors recommended that the definition of forest degradation should be consistent with established definitions such as those employed by the UNFCCC, the Kyoto Protocol, the Marrakesh Accords, and other widely used definitions such as those adopted by the FAO. For the purposes of greenhouse gas policy and accounting they proposed a number of principles as a basis for defining forest degradation, including that:

- degradation is related to direct human-induced changes in carbon stock,
- accounting for emissions from degradation include carbon stock changes in all relevant pools and emissions of non-CO_2 greenhouse gases,
- the definition should rely on quantitative, objective standards, be unambiguous and support inventory and reporting in a rigorous, verifiable, and transparent manner,
- the definition be capable of being assessed using technically feasible methodological options for estimating emissions that are easy to apply and consistent across a wide range of biomes and relevant vegetation types.

For consistency in accounting, forest degradation should not be confused with deforestation. If human activities involve a long-term reduction in forest cover, height, and area that would result in reclassification of land as non-forest then this is considered deforestation.

Penman *et al*. (2003) also raised concerns about *unbalanced* or *incomplete* accounting, where emissions and removals from relevant pools and lands are not all reported or recovery in parts of the forest estate are not included in the accounting framework.

As a starting point for analysis of carbon cycling and climate change mitigation, it is proposed that 'degradation' be considered a human-induced process that results in a long-term reduction in forest carbon stocks. Reduction in forest canopy cover is not sufficient for the land to no longer be defined as forest. The key challenges are defining 'long-term', the extent of reduction in carbon stocks and the area to be considered subject to 'degradation'.

A framework for assessing forest carbon dynamics

Accounting and managing for greenhouse gas emissions in forests presents new challenges for forest owners and managers. Carbon pools specified for greenhouse accounting are: above and below ground living biomass, dead wood, litter and soil organic matter (IPCC 2006) and methods for assessing greenhouse gas emissions resulting from deforestation are relatively well-defined (IPCC 2006). The recommended approach combines 'activity' data (the area converted from forest to other forms of land cover) with associated emissions factors, generally derived from remote sensing and ground measurements. Increasing levels of certainty in estimates are provided with more complex methods, regional specificity of model and emission factor parameters and the spatial resolution and accuracy extent of activity data.

It is also possible to assess emissions resulting from forest degradation using these approaches, although low or small thresholds between one state and another may require higher-resolution remote sensing with continuous spatial coverage; higher intensity sampling systems, or more detailed and comprehensive activity reporting systems. In order to assess whether long-term reductions are occurring, continued monitoring and measurements are required through time (Penman *et al*. 2003). The scale of any impacts is also important in determining whether the activity is worthy of policy attention as a climate mitigation measure.

Changes in carbon stock can be estimated at various scales and levels of accuracy depending on investment in remote sensing and inventory and data available within countries. According to IPCC Guidelines, carbon stocks in undisturbed natural forests are generally considered to have a carbon balance of zero.

Using these different principles, a framework for assessing GHG emissions associated with forest degradation would involve:

- Assessment of the forest area subjected to a potentially degrading human activity.
- Determining the extent of reduction in carbon stock.
- Determining the length of time over which carbon stocks are reduced.
- Comprehensive accounting to include lands subject to past or present forms of the activity.
- Providing for balanced accounting and assessing recovery of carbon stocks.

The extent of reduction in carbon stock can be estimated directly, or using emissions factors associated with indirect estimates of different activities. For example, reductions in carbon stock due to timber harvesting are often assessed using data on timber removals as a starting point, because these are more readily available. Change in practices or measures to reduce greenhouse gas emissions would compare the stock change in one period (the baseline) with the change in the target period for emissions reductions.

Case studies from the Oceania region are now considered to illustrate different issues associated with accounting for greenhouse gas emissions associated with forest management or forest degradation. Australia and PNG are near neighbours with quite different ecologies, histories of human cultural development, European settlement, land uses, economic development and current land use and forest management. In the international arrangements for climate mitigation forest degradation is likely to be treated quite differently in the two countries. Australia, as an Annex I party in the Kyoto Protocol will need to consider any application of forest degradation in context of comprehensive accounting under 'forest management'. PNG as a developing country, may be able to take an activity-based approach. Nevertheless, they provide an interesting basis for comparing the potential application of forest management and forest degradation in greenhouse gas accounting and reporting arrangements and emission reduction targets and it has been proposed that they become more deliberately linked in greenhouse gas emission reduction objectives (Garnaut 2008).

Australia

Forests cover 149 million hectares (24% of the land area), 147 million hectares of native forests and nearly 2 million hectares of forest plantations (MIG 2008). About 70% of Australian forests are under private ownership or management through long-term lease arrangements. There has been a long history of forest use, by aborigines through extensive use of fire which shaped the Australian landscape and its vegetation composition and structure and more recently by Europeans. Significant areas of native forests have been converted to agriculture. Clearing of forests and the introduction of predators has resulted in extensive habitat and species loss.

Plantations established prior to 1990 in Australia were largely softwood and under public ownership and mostly established on areas converted from native forests. Plantations established after 1990 were largely planted on cleared land with private capital, primarily for short rotation pulpwood crops. Plantations now supply over 70% of the current timber harvest of 28.5 M m^3 per year.

There has been a history of controversy over forest use (Dargarvel, 1995; Lindenmayer and Franklin, 2003). In the early 1900s there was considerable debate about the extent of forest protection versus conversion to agriculture in the interest of economic development. Following World War II, accelerated harvesting in native forest to meet demand for post-war housing construction, large-scale conversion of native forest to exotic pine plantations and the development of export woodchip markets, which resulted in more intensive harvesting in some native forests, created a strong reaction in parts of the community at a time when environmental, cultural and recreational values of forests were becoming increasingly important in society, particularly those in urban areas.

The 1992 National Forest Policy Statement sets the current policy framework for forest management. Regional Forest Agreements (RFAs) for 10 regions were signed between state and federal governments with specific targets for conservation reserves including provision for protection of rare or threatened species and ecosystems, old growth forests, wilderness and cultural values (Davey *et al.* 2003). Subsequent land use decisions by individual state governments have further increased the area of forest in conservation reserves (MIG 2008).

Carbon dynamics in Australian forests are determined by a variety of processes: conversion to agriculture or urban development, plantations and other forms of afforestation, timber harvesting, drought and fire. Recent analysis indicates that there are emissions in some forest types due to land clearing, dieback and fire and gains in others due to regrowth and thickening, with the net carbon balance of forests being broadly maintained (MIG 2008, Criterion 5). This paper focuses on native forest harvesting and fire as two processes that could potentially be defined as forest degradation.

Native forest harvesting

Australian native forests supplied considerable quantities of timber for construction and furniture in the early years of European settlement. Silvicultural and management systems developed to provide for regeneration and long-term sustained timber yield and a variety of other management objectives (McKinnell *et al.* 1991, Bauhus 1999). Harvesting increased significantly from 1950-1990 to meet the needs of post-war construction and developing pulp and paper industries. Currently, about 113 M ha (76% of the native forest area) has no legal restriction on wood production. However, only around 15-20 M ha is likely to have suitable species and proximity to markets to make commercial timber harvesting economically viable. The area of public forests zoned for multiple-use and available for harvesting declined from 11.4 M ha in 2000 to 9.4 M ha in 2006 and the area of public nature conservation reserves increased from 21.5 M ha to about 23 M ha over the same period, as a result of processes to develop the conservation reserve system (MIG 2008).

The carbon dynamics in intact native forests are complex. Forests vary widely in condition, with significant areas of old growth, with large trees and high carbon stocks (Raison *et al.* 2003). Others are in a regrowth condition or mixed-age class following past disturbance. There has been no systematic, repeated field-based inventory of Australian native forests as there has been in other jurisdictions (eg. Woodbury *et al.* 2007) and forest carbon dynamics are generally not well understood. Past timber harvesting may mean that carbon stocks in native forests are below their potential (Roxburgh et al 2006) as is the case in other regions (Brown *et al.* 1997), although light selection harvesting may actually result in higher total forest carbon stock than in unharvested areas (Ranatunga *et al.* 2008). It has been suggested that native forest timber harvesting be considered a form of forest degradation because it reduces forest carbon stocks relative to ‘its natural carbon carrying capacity’ (Mackey *et al.* 2008).

Timber harvesting rates are a function of the area of forest available for harvesting, estimated growth and market conditions. Native forest harvest levels have declined by about 10% from harvest levels during the 1990s to an average of about 9.2 M m^3 over the last 5 years (ABARE 2009).

Using timber removals as a basis for activity assessment involves assessing the quantity of carbon removed in harvest and emissions from harvesting slash (about 0.9 times the timber removed, DCC 2009). Using a simple multiplication, harvest is estimated to result in a reduction in native forest carbon stocks of 5.68 M tonnes, equivalent to about 20.8 M tonnes of CO_2. This does not allow for emissions from harvesting slash actually occurring over time through decay and a proportion of the timber removed being added to the wood products pool, not the atmosphere. Emissions from burning firewood are estimated to be about 10 M tonnes of CO_2 (MIG 2008). Regrowth native forests were estimated to take up about 43.5 Mt CO_2 per year in 2005 (MIG 2008) resulting in a net increase in carbon stocks in managed native forests of about 13 M tonnes of CO_2 per year.

Consequently, at a national scale, native forest timber harvesting and associated regeneration in managed native forests is not resulting in a long-term decline in forest carbon stocks compared with any recent baseline period. Carbon emissions associated with timber harvesting in Australian native forests are actually likely to have declined by about 10% compared to a 1990 or 2000 baseline timber removals from native forest have declined over the last 10 years. In addition, net emissions are likely to have reduced further because harvesting has shifted from old growth and mature forests to regrowth forests (with reduced emissions from harvesting slash). The estimated carbon uptake in regrowth also does not include sequestration in a substantial area of regrowth forests allocated to new conservation reserves and not now considered part of the 'managed' forest estate.

At a local scale, carbon stock reductions may be occurring in situations where old growth or mature forest is being converted to regrowth through clearfelling or where more intensive management of regrowth is being applied through thinning and shorter rotations.

Large-scale wildfire

Fire has been part of the global environment since the evolution of terrestrial plants and it has a variety of direct and indirect impacts on the global climate systems (Bowman *et al.* 2009). Fire is a widespread feature of the Australian landscape and much of the vegetation is adapted to or even dependent on fire and fire-related disturbances for regeneration and survival (Bradstock *et al.* 2002). Wildfires have significant impacts on life, health, property, infrastructure and primary production systems (Whelan *et al.* 2006) and fire affects nutrient cycling and availability, forest productivity, vegetation composition, wildlife habitat, soil biota and hydrological functioning. Large-scale wildfires can occur as a result of deliberate (arson) or unintentional (eg. powerlines) human activities or naturally through lightning strikes. Prescribed fire is widely used as a management tool, although this practice is not without controversy and the rate of burning in eastern Australia has declined in recent years after peaking in the early 1980s (Tolhurst 2003, quoted in Attiwill and Adams 2008). Fires are more intense and severe in southern regions with Mediterranean climate (hot dry summers and cool wet winters).

The incidence and severity of wildfire varies considerably from year to year and decade to decade, depending on climatic conditions, vegetation type, fuel loads and human actions and there may be prolonged periods (75-150 years) between 'stand replacing' fires (McCarthy *et al.* 1999). The area of forest impacted by fire is not well-monitored and fires vary greatly in extent, intensity and severity. 'Emissions factors' associated with different fire intensities are also uncertain. Some is emitted directly as CO_2 but smoke is a complex mix of organic compounds, particulates and carbon monoxide. A considerable amount of wood remains in the forest, some is reduced to charcoal, that may have long-term stability in the soil and dead standing trees eventually decompose over time.

In south eastern Australia, there has been a relatively high incidence of large scale wildfires since 2000, with major events in the summers of 2002-03, 2006-07 and in February 2009 partly as a result of prolonged period of below average rainfall. Over 2.6 M ha have been burnt in these events with significant consequences for forest carbon stocks. Estimates of carbon dioxide emissions associated with these fires range from 40 Mt for the first (MIG 2008) to 600 Mt for both (Attiwill and Adams 2008). The most recent events on 7 February 2009 that resulted in the deaths of 173 people and the loss of over 2,000 homes may have resulted in carbon emissions of over 50-100 Mt of CO_2.

If the fire interval is sufficiently infrequent, this emitted carbon will subsequently be sequestered in regrowth following the fires. However, successive fires in the same area may mean a permanent reduction in forest carbon stocks. Higher fire frequency can lead to shifts in vegetation composition and reduced forest carbon stocks. Regular prescribed fire can reduce the intensity of wildfires.

Given that our understanding of carbon dynamics due to fire is uncertain and they are caused by a mix of natural forces and human actions it is difficult, at present, to class fire as 'forest degradation'. However, under climate change scenario, the frequency of severe fire weather days is projected to increase over the next 20-40 years (Hennessy 2007) and the future interaction of human-induced and natural processes may result in the carbon impacts of increased fire frequency being regarded as forest degradation.

Papua New Guinea

Papua New Guinea lies to the north of Australia between the equator and latitude 12^{o} south. There are a wide variety of environments and forests are characterised by high species diversity. Tropical forests and freshwater wetlands of PNG are considered to be of similar biological and conservation importance to the Amazon and Congo Basins (Collins *et al.* 1991, Alcorn 1993). Human societies in PNG are also highly diverse, with over 700 different language groups and a large number of different cultural and ethnic groups in the population of about 6 million people. In coastal and lowland regions most communities have practiced some form of shifting cultivation within a fairly well-defined area of secondary vegetation, with a variable expanse of primary forest separating the gardening zones of neighbouring groups. Population growth is high (2.3%).

There are five primary drivers of forest cover change in PNG: subsistence agriculture, timber harvesting, fire, plantation conversion and mining. Conversion to intensive agriculture has been relatively limited. For example, about 120,000 ha of oil palm plantation has been established after over 30 years of development and not all has been a result of forest conversion. Fire has been shaping PNG's vegetation patterns through thousands of years of human settlement (Johns 1989, 1990; Haberle *et al.* 2001). If the interval is not too frequent, forests generally recover from fire and the structure of forests has in some parts been determined by previous fires. Mining has locally significant impacts on forest cover, particular in Western Province, where the siltation from the Ok Tedi mine is continuing to cause death of riparian forest. These two case studies focus on the potentially degrading activities of shifting cultivation and fire.

Shifting cultivation

Land in PNG has been used intensively by humans for subsistence agriculture production for thousands of years with landowners operating on a cycle of clearance, cultivation and regeneration in which the key variable is the period of time for which the land is cultivated and left to fallow (Ruthenberg 1980).

It has been suggested in a recent analysis of remotely sensed data that shifting cultivation is resulting deforestation in PNG as a result of increasing rural populations and encroachment of shifting agriculture into primary forests (Shearman *et al.* 2008). However, other field surveys indicated that, while land use is intensifying as a result of increased population pressure, this is largely occurring on land previously used for subsistence agriculture and not at the expense of primary forests (McAlpine and Freyne 2001). It is actually very hard work for local farmers to convert primary forests to areas suitable for cultivation and observations suggest that farmers prefer to maintain a system of rotation in which forest fallows or areas of 'secondary forest' are cleared for cultivation at intervals of at least 10 years.

This is generally supported by Allen *et al.* (2001) who found that about 11 M ha of land was being used by local farmers, with 50% being left in fallow for periods of more than 15 years, 43% for periods of 5-15 years and only 7% for less than 5 years. Most agriculture in regions with population densities exceeding 100 persons per square kilometre already had non-forest forms of fallow when they were surveyed in the early 1990s. Tall secondary forest was the typical fallow vegetation cleared for cultivation on 56% of the land used by local farmers with just 0.11 M ha considered to be at risk of 'degradation'. Reductions in fallow periods (and consequently long-term reductions in carbon stocks) may be occurring but the evidence suggests that this is not happening on a large-scale. Only 12 locations involved previously unused primary forest was being cleared for cultivation.

Consequently, shifting cultivation is not likely to be resulting in long-term carbon stock reductions on a significant scale compared with any likely recent baseline period. It therefore is unlikely to be considered a cause of forest degradation. Given that this practice is a complex cycle of clearance, cultivation followed by secondary forest fallow, repeated intensive monitoring will be required to detect reduced fallow periods or permanent conversion of secondary forest to intensive agriculture. The emission reduction benefits of reducing these activities are unlikely to justify the cost of monitoring.

Timber harvesting in native forests

Almost all land in PNG is under customary or tribal ownership. Forests have been seen as basis for economic development for some time and industry development policies in the 1960's and 70's initially favoured domestic processing. However, these were generally small-scale and aimed at providing building materials to the local market. The government began to encourage landowner involvement in log exports in the 1980s and PNG has become the second largest exporter of logs in the Asia Pacific region, after Malaysia. The volume exported fluctuates considerably from year to year (Fig. 1). There is a small but growing export market for timber produced from community-based management and small-scale certified operations using portable sawmills (Bun and Scheyvens 2007).

Under the National Forest Plan, 11.9 million ha of land is identified as the production forest estate. Most of the forest subject to timber harvesting is allocated to processors and exporters through timber rights purchase and management agreements negotiated between the government and landowner groups and approved by the National Forest Board. The annual harvest set for each area assumes that the return interval between harvests is 35 years (Lowman and Nicholls 1994).

Harvesting is selective and the intensity of fellings varies widely with the density of merchantable species and the market requirements and skill of the forest operator. Some areas are permanently converted due to roading or inadequate regeneration. Unlike in Australia, there is some recent data on the impacts of harvesting on carbon stocks that can be used to assess the carbon impacts of selective harvesting. Average timber removals are 10-20 m^3/ha (Keenan *et al*. 2005). Using a figure of 15 m^3/ha with the reported annual log export volume results in an estimate of the total area impacted by selective harvesting of about 3.2 M ha. This is similar to previously reported estimates (McAlpine and Quigley 1998) but less than the 3.8 M ha Shearman *et al*. (2008) suggested had been impacted by logging in their recent remote sensing study.

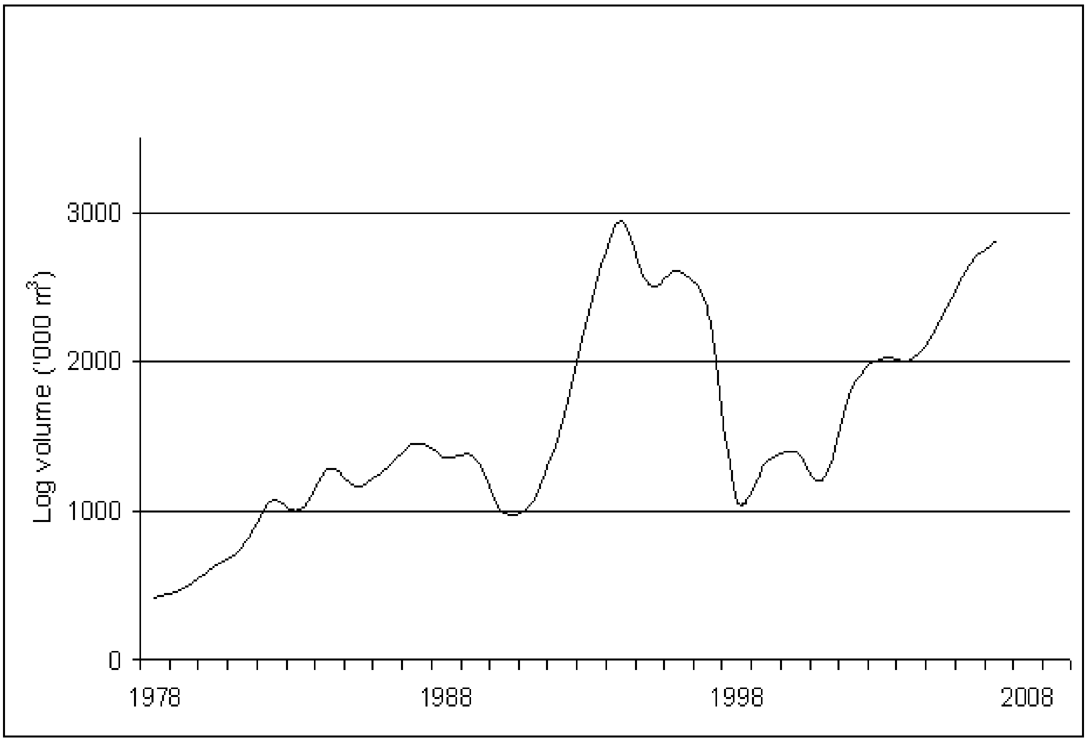

Figure 1: Raw log exports from PNG, 1978–2007.

Source: Bank of Papua New Guinea, Quarterly Economic Bulletins.

Analysis of 125 permanent sample plots across the country (Fox et al 2009) indicated that the average carbon density in above-ground live biomass (AGLB) in undisturbed lowland primary forest was 125.3 MgC/ha (SD 28, 11 plots) and 83.8 MgC/ha (SD 24.2, 114 plots) in AGLB logged forest. Thus, logging resulted in a 33% reduction in carbon in above-ground biomass. Applying this stock reduction to the harvested area associated with timber removals gives average CO_2 emissions associated with timber harvesting of about 26 Mt CO_2 per year over the 10 years from 1999 to 2008 (including an allowance for total loss of carbon stock on areas subject to roading and conversion to gardens).

However, it is the long-term fate of these stands that will determine whether the impact on carbon stock is long term and can be considered forest degradation for carbon accounting purposes. Of 89 plots with measurement periods longer than about 5 years 68 (or 76.4%) showed an increase in basal area over the period (Yosi *et al.* 2009) and the average rate of carbon accumulation in AGLB in harvested forest was 2 Mg C $ha^{-1}yr^{-1}$ (SD 2.0, Fox *et al.* 2009b). Applying these figures to the cumulative area harvested indicates that the above emissions are offset by removals of 12.8 Mt CO_2 per year.

Thus, timber harvesting is resulting in a reduction in forest carbon stocks, but the net impact is offset to some extent by removals in regrowth in a high proportion of harvested stands. Whether this stock reduction is long-term will depend on future activities in the forest. This will depend on stocking and rate of growth of merchantable species, the extent and location of the forest, future market conditions and, possibly, the value of avoided greenhouse emissions. Some accessible harvested forests are already being subjected to further cutting, either by larger companies or small-scale sawmilling. Establishing the baseline is possible given the available data but the actual baseline will vary considerably depending on the period chosen. Accounting for reduced emissions associated with changes in forest management practices and incorporating this into a payment mechanism such as REDD through ceasing or reducing the impacts of harvesting (Putz *et al.* 2008) will require both harvesting emissions . Ongoing monitoring will be required to assess the whether the stock reduction is long-term and degradation is occurring.

Discussion

This analysis presents four different types of disturbances occurring in natural forests that might be considered to meet the definition of forest degradation:

1. In the case of timber harvesting in Australian native forests, this is largely occurring in regrowth forests. Greenhouse gas emissions from harvesting are generally exceeded by sequestration in forest regrowth and there is a net increase in carbon stocks. Therefore this activity does not meet the proposed definition of degradation for greenhouse accounting purposes. The capacity to monitor the area affected by harvesting has improved considerably in Australia in recent years. Improved estimates of carbon dynamics, including the generation and fate of harvest slash and rates of forest growth and carbon sequestration following harvesting are still key areas of uncertainty in accounting for greenhouse gas emissions associated with this activity.
2. Wildfire in Australian forests is not currently considered a human-induced activity and carbon stocks reductions by fire are considered balanced by subsequent regrowth. However, the recent high frequency of intense and destructive wildfires in south eastern Australia, their human origins (in some cases) and the projected higher fire incidence under some climate change scenarios, raises the prospect that such activities could, in the future, be classed as forest degradation. Estimates of greenhouse gas emissions associated with wildfire and prescribed fires are still highly uncertain and improved information on the dynamics of forest carbon stocks following fire, including impacts on soil carbon, charcoal and the fate of organic material subject to incomplete combustion (Attiwill and Adams 2008) are required to better inform greenhouse gas inventories.
3. Shifting cultivation in Papua New Guinea primarily occurs in areas that have previously been converted to gardens and left to regenerate as secondary forest fallows. It has been proposed primarily as a cause of deforestation but may also result in forest degradation if there is a long-term reduction in the length of secondary forest fallow. However, there is little evidence that this is occurring. Repeated observation through remote sensing and field surveys is required to detect degradation associated changes in the fallow period.
4. Selective timber harvesting in PNG is occurring primarily in intact primary forest and results in reduction in forest carbon stocks. Given that these are in proposed production forest areas, there are likely to be repeated harvests and the stock reduction is likely to be long-term and therefore the activity can be classed as forest degradation. However, balanced accounting would include regrowth from previous timber harvests, reducing the overall impact of harvesting. The rate of harvesting has varied considerably over time and the net effect of harvesting will depend very much on the setting of the baseline period. Ongoing monitoring through remote sensing and field survey will also be required to effectively assess the trajectory of forest carbon dynamics following selective harvesting.

The establishment of a sound baseline using historical remote sensing data or aerial photographs requires considerable local knowledge to properly interpret forest condition prior to the introduction of the potentially degrading activity and to effectively monitoring sound estimates of emissions from forest degradation require effective monitoring of forest growth and dynamics following different types of disturbance.

In Australia, the interaction of timber harvesting and fire impacts is likely to become an increasingly important issue under future forest management and climate change scenarios (see analysis for forests in the

USA by Hurteau *et al*. 2008). Maintaining variability in forest structure and understorey species can maintain carbon stocks and provide wildlife habitat benefits. Any decision to cease harvesting for climate change mitigation objectives would need to consider these interactions as well as the social and economic consequences and the potential leakage of emissions to other forests. Analysis of forest management effects on carbon stocks also should not stop at the forest boundary. The recent Fourth Assessment Report of the IPCC noted that carbon storage in wood products, replacing energy intensive building materials and fossil fuel emissions using biofuels from wood can increase forest carbon benefits. We need to utilise extracted timber as efficiently as possible to further increase carbon stocks in wood products and replace emissions from fossil fuels. The optimal solution to forest carbon management needs to consider the whole carbon lifecycle (Lindner, M. *et al*. 2008).

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