



Effects of Forest Certification on Biodiversity

Marijke van Kuijk, Francis E. Putz & Roderick Zagt





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Foreword

The majority of the world's terrestrial species is found in forests and most of that is in the tropics. Forests also provide livelihoods to millions of people, many of whom are poor. The carbon stored in forests and the other ecosystem services they provide are of local, regional, and global concern. Given a global deforestation rate of about 13 million ha per year, and an unknown but considerable higher rate of tropical forest degradation, increasing efforts at maintaining forests and their biodiversity through improved forest management should be an important global priority (see Rametsteiner and Simula 2003 for references).

Throughout the world, forestry operations have substantial impacts on biodiversity. Some of these impacts are intended, as when silvicultural treatments are applied to maintain or enhance the stocking and growth of commercial species, but others have undesirable environmental effects. Efforts at decreasing the deleterious impacts of forest management focus on logging because, of the many possible silvicultural interventions, logging is the most common and usually the most detrimental. Forest certification is increasingly viewed as a way to diminish these detrimental impacts and to make harvesting more sustainable from ecological, economic, and social points of view.

The Planbureau voor de Leefomgeving (PBL, or Netherlands Environmental Assessment Agency) requested Tropenbos International to conduct a literature study to evaluate the degree to which forest certification impacts on the biodiversity of temperate, boreal, and tropical forests. This question is exciting, because of the important role assigned to forest management certification as a mechanism to conserve forest biodiversity, and the challenges related to assessing and interpreting measuring biodiversity impacts. To date, little comparative research has been carried out into the biodiversity of forests prior to and post certification, or of certified and non certified forests. Apart from the value of assessing the impact of forest management certification, for agencies such as PBL it is relevant to be able to advise governments and other relevant stakeholders on policies and instruments that produce the best results in terms of biodiversity conservation, and under what conditions.

In this report we assess the effects of forest certification on biodiversity by discussing what biodiversity is, how it is measured, what sustainable forest management is (or should be), and by analyzing the available scientific literature on the effects of forest management activities and certification on biodiversity. We conclude with an evaluation and discussion of our findings. It is a companion to a report by Probos on the practical aspects the effects of certification on biodiversity (Jansen and van Benthem 2009).

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Summary

Does forest certification work for biodiversity conservation?

Forest certification is widely seen as an important component of strategies for conserving the world's forests. During the 1990s concern about the loss of biodiversity in logged forests was a key driver behind the emergence of forest certification. It was thought that production forest could play a bigger part in conserving nature by adhering to a strict and widely agreed forest management standard that considers the effects of logging and other forest management activities on biodiversity. Since the introduction of forest certification more than 300 million hectares of forest have been certified under a variety of schemes, the majority of which are located in temperate and boreal areas. Less than 20 million hectares are in the tropics, mostly certified by the Forest Stewardship Council (FSC).

Although interest in forest certification has waxed and waned, it remains a cornerstone of forest policies. But does it work? As more than 15 years have passed since the first certificate was issued, it should be possible to evaluate the effectiveness of certified forest management by comparing the conservation performance of certified forests with non-certified forests.

A literature study on this subject was conducted to assess the scientific evidence for an effect of certified forest management on biodiversity in tropical, temperate and boreal forests. Within the wealth of literature discussing the impact of logging on plants and animals, this study focused on studies which compared certified with 'conventional' forestry practices. Forced in part by the near absence of studies directly addressing the effects of certified forest management on biodiversity, the review also addressed studies examining effects of 'good forest management practices' that are often associated with certification:

- reduced-impact logging (RIL)
- establishment of riparian buffer zones
- green tree retention (GTR)
- establishment of corridors
- protected areas within forest management units
- identifying High Conservation Value Forests (HCVFs)

A total of 67 studies have been reviewed to evaluate the hypothesis that forest biodiversity in well-managed forest management units is higher or more intact than in otherwise similar but conventionally managed areas.

Complexity

The study has revealed the difficulty of providing a clear answer to this hypothesis. In most certified forests, even though forest managers collect biodiversity information, systematic collection of information needed to assess the effects of management on biodiversity does not take place. Data from non-certified forests, which are needed to assess the added value of certification, are even harder to find. The scientific community has not yet risen to the challenge of providing evidence of the effects of certified forest management on a comprehensive scale. Studies focus on different species, use different concepts of measuring biodiversity and a variety of field protocols and often do not address the temporal and spatial scales appropriate to forest ecosystems and forestry.

Even though this is regrettable, it does not come as a surprise, given the complexity underlying the concept of 'biodiversity' – even its simplest definition as species richness and abundance. Different species, even related ones, respond in different ways to the same management activity, and require different research protocols. Moreover, the short-term effects of logging may be very different from the long-term effects. Conclusions drawn about the changes in a simple and convenient metric, such as the number of species present in a forest, may fail to reveal underlying shifts from forest specialists to habitat generalists.

Similarly, 'certified forest management' is a complex topic. Forests differ from place to place, and so management practices vary. Differences in logging intensity, logging pattern and timing, the size and variety of species harvested, extraction methods and post-harvest activities all contribute to different responses by plants and animals.

Conclusions

In the face of these difficulties and uncertainties, the conclusions drawn from the literature study can only be tentative. Only a handful of studies, all in a certified forest in Sabah, have directly assessed the effects of certified forest management on a number of plants and animals. They showed that populations of endangered animals increased.

The main conclusion is that in spite of a very large variety in responses between species, **the forest management practices associated with forest certification appear to benefit biodiversity in managed forests.**

There is evidence that 'good forest management practices' associated with forest certification are beneficial for the conservation of forest biodiversity across species groups and across geographical regions – in spite of variability in responses between species and the existence of exceptions. Negative effects of logging on forest species are reduced by reduced-impact logging since it causes less damage to the forest than conventional logging techniques. Riparian zones offer specific habitat characteristics of which many aquatic, semi-aquatic, and terrestrial species are dependent for many, if not all, stages of their life cycles. Protecting these zones against logging damage thus contributes to species preservation although the

extent to which species benefit from these protected zones depends on zone width and several other factors. Green tree retention in clear cuts maintains some of the habitats present before logging, on which many species depend. Thus, when compared to total clear cuts, retention trees provide a benefit to many species but the magnitude of the benefit depends on the type and the number of retention trees. Corridors provide shelter to many species and provide links between otherwise isolated patches of remaining habitat. Therefore, they benefit many species in intensively logged areas. Size, shape, and connectivity of these corridors determine their effectiveness in species conservation. It seems logical that protected areas within logging units and HCVF protect many species from the negative impacts of logging, but we found few data to support this conclusion.

This conclusion is in agreement with information gathered from discussions with certifiers and forest managers, and with experiences in the field, and confirms the findings of certification impact reviews based on required improvements in forest management due to certification. Despite the apparent differences in the rigour with which biodiversity concerns are addressed under different certification systems, the planning, supervision and basic good management practices required by all of them serve to mitigate many of the harmful environmental impacts of logging and other forest management activities. Similarly, these studies confirm that despite their better performance, certified forests are not fully equivalent to undisturbed or primary forests in terms of biodiversity.

This main conclusion must be qualified by a number of additional observations:

- There is a very high variation, *both* in forest management practices associated with certification *and* in responses between and even within species;
- there is little quantitative evidence about the long-term effects of certified forest management on biodiversity;
- there are few data on which to base the conclusion that certified forest management is *sustainable* in terms of biodiversity conservation at the level of populations and communities – we simply don't know.

Recommendations

The review also shows that the impacts of certified forest management on biodiversity cannot be assessed without a clear idea of the relative importance of species and of management objectives. This leads to several recommendations.

- As different species may be valued differently by different stakeholders – based on considerations of rarity, vulnerability, endemism, distinctness, economic usefulness, potential as a pest, religious and spiritual value, and many other considerations – formulating appropriate functions of production forests in conserving biodiversity requires debate and negotiation at the local level (but without dismissing global interests).
- The results of these discussions must be translated into practical management activities for achieving specific, measurable biodiversity objectives. These

must be subject to periodic revision to accommodate changes in value perception and in the state of biodiversity in the forest.

- To further inform the trade-offs between biodiversity and the social and economic interests of forest management accepted by certifiers, scientists will have to provide quantitative, field-based evidence of species responses to forest management practices, and to propose modifications if that is required.
- Finally, biodiversity monitoring and audits of certified forest management should focus on these practical management activities and objectives rather than on general, unspecified biodiversity goals, which are almost impossible to measure and, if they can be measured, hard to interpret.

The challenge for forest managers, certifiers and biodiversity researchers will be to promote forest certification from a credible proposition to a demonstrated asset in the suite of instruments available for forest biodiversity conservation.



1

Introduction and background of the study

Despite widespread concern, policy reforms and a variety of initiatives, global rates of biodiversity loss remain alarming. On the 2008 Red List, for example, one out of every four mammal species (IUCN *et al.* 2008a), one out of eight bird species (Birdlife International 2009), and one out of three amphibian species were reported to be endangered (IUCN *et al.* 2008b). More than one third of the European fresh water fish are threatened with extinction (Kottelat and Freyhof 2007). In Asia, 70% of monkey species are considered 'threatened' (IUCN *et al.* 2008a). Important and interrelated causes of biodiversity loss include increasing human population, deforestation, forest fires, pollution, and climate change (MEA 2005).

During the World Summit on Sustainable Development in 2002 in Johannesburg, more than 200 countries agreed to substantially reduce the decline in biodiversity by 2010. In October 2008 during the World Conservation Congress in Barcelona it became apparent that this goal will not be reached. According to experts attending the Congress, only 16 countries have adopted the policy measures needed to reach this goal. Experts also reported that heads of environmental ministries typically have less input and influence than ministers of economic affairs, which makes it especially difficult to use national policies to reach biodiversity goals. For instance, it is expected that for most of Europe, national goals will not be reached before 2050 (www.iucn.org).

The majority of terrestrial biodiversity is found in forests, which cover about one third of the Earth (FAO 2005). Each year about 7 million ha of forest disappear, mostly in the Amazon but also in the Congo Basin and Asia (FAO 2005). In May 2008, 59 countries at the UNCBD in Bonn agreed that the decline in forest area and thus the decline in biodiversity should be stopped by 2020.

Apart from policy initiatives to stem the loss of forest biodiversity, forest certification has emerged as a practical, market-based instrument to stimulate wise management of forests. In this way, individual consumers can contribute to biodiversity maintenance by being selective in their choice of wood products. By purchasing wood from certified well-managed forests, they could help arrest the loss of biodiversity in managed forests.

Efforts from Governments to exclude timber that is not produced in a responsible manner from the market have lagged behind. To help the environment at home and abroad, the Dutch government set the goal that from 2010 onwards all national

government procurement¹ will be from sustainable sources, i.e. certified timber in the case of timber. By 2011, the target is that at least 50 % of the wood on Dutch market will be from sustainable sources. Earlier goals to have timber in the Dutch market certified by 1996 (100 %, RTR 1991), 2000 (100 % PIN 1995) and 2005 (25 %, BBI 2002) were not achieved. This was due in part to disagreement and confusion about the degree to which certification systems met the Dutch sustainability criteria. Different certification systems are currently being evaluated for their sustainability by the Timber Procurement Assessment Committee (TPAC, Dutch Ministry of VROM). At the time of writing, wood certified by FSC International, PEFC Finland and PEFC Germany had been approved by TPAC as “sustainable”. The recent ‘Beleidsprogramma Biodiversiteit 2008-2011’ (Biodiversity Program) of the Dutch government, highlights the role of sustainable supply chains in the effort to prevent a further loss of biodiversity. Activities were proposed that must lead to achieve the target of 100% sustainable government procurement by 2011.

Apart from the discussion about targets and admissibility of certification systems, this requires an understanding of the effectiveness of forest certification in reducing biodiversity loss in managed forests – which is as yet unanswered. As part of its mandate to monitor the Biodiversity Program, the Planbureau voor de Leefomgeving (PBL, or Netherlands Environmental Assessment Agency) commissioned a study to assess the effects of timber certification on biodiversity.



¹ i.e., 100% of central Government procurement, 50% of procurement by provinces and water boards and 75% by municipalities.



2

Approach

The Terms of Reference by PBL ask for an assessment of “*the measurable effect of specific forms of sustainable forest management (SFM) on biodiversity in forests in various climate zones*”, as compared with conventional forest management. We address this charge by conducting a literature survey on the biodiversity benefits of various forest management certification systems in operation around the world.

2.1 Interpretation of biodiversity in this report

Biodiversity refers to the natural variety and the physical organization or pattern of the variability among living organisms (Putz *et al.* 2000). These authors distinguish five hierarchical components of biodiversity, namely at the gene, species or population, community, ecosystem and landscape levels, each of which has structural, functional and compositional attributes (see also section 4.2). Logging and sustainable forest management interact in complex ways with these components and attributes of biodiversity. In the praxis of sustainable forest management, however, the term biodiversity is pragmatically interpreted as the compositional attribute of communities, namely species richness, or of species, namely species abundance. In this report we therefore adopt this as the interpretation of biodiversity. The other attributes of biodiversity are critical in understanding the effects of forest management, but they require dedicated research² and are not routinely determined in monitoring schemes in forestry. We mention them when we considered it relevant.

As a second limitation to the interpretation of the term biodiversity, we focused on the effects of forest management on forest biodiversity, as opposed to ‘total’ biodiversity. In this way we deal with the commonly observed effect of increasing or maintaining biodiversity following logging (e.g., Cannon *et al.* 1998, Zagt *et al.* 2003), resulting from an influx of non-forest species into new and open habitats created by logging. Even though biodiversity may remain the same or even increase, at least temporally, we feel that conclusions based on those observations would detract from the underlying concern about poor logging practices, i.e. the loss of typical forest species. This issue opens the discussion on what is a forest species, what is ‘typical’ forest biodiversity to be protected, and eventually, what is the relative value of different species. We touch upon this discussion again in section 8.3.1.

² Examples include studies on the impact of certification on landscape level biodiversity attributes by Hughell and Butterfield 2008; de Koning (2008 in Karmann and Smith 2009), on the effect of certification on conversion rates of forests into other land uses.

Even though the effects of forest management on red-listed, threatened and rare species are important, we did not place a special emphasis on those taxa, but rather included all plant and animal taxa covered within the literature we reviewed. Most of the studies described single species responses or species group responses to management activities.

2.2 Interpretation of good management practices

Answering the question about the effects of forest certification on biodiversity ideally relies on direct measurement of species composition and abundance across spatially replicated, ecologically similar, certified and non-certified forests with similar histories, based on a time series of assessments both pre- and post-certification. Studies taking this approach are virtually absent³. We, therefore, supplement the direct evidence on biodiversity impacts with information on the effects of good forest management practices most likely to be adopted as components of certified forest management, on the richness and abundance of species.

Note that implementation of such management activities does not provide direct evidence for the desired effect of certified forest management on biodiversity, but makes such effects likely.

Among the wide variety of 'good management practices' we focus on the following which we considered most relevant for this study:

- reduced-impact logging (RIL)
- establishment of riparian buffer zones
- green tree retention (GTR)
- establishment of corridors
- protected areas within forest management units: demarcating and protecting sensitive and representative areas in areas that would be otherwise harvested
- identifying High Conservation Value Forests (HCVFs) and designating special methods for their management – a practice strongly associated with certification.

The literature on the effects of forest management on biodiversity is primarily focused on logging, but there are other silvicultural treatments that also warrant attention. Among the many techniques silviculturalists can use for forest management, soil scarification and controlled burns are most often employed in temperate and boreal forests. Liana (=woody vine) cutting and liberation of future crop trees (FCTs) from competition are the most common in the tropics. Whereas these and other treatments are actually employed in many temperate/boreal forests, other than in certified areas, silvicultural treatments in the tropics are actually applied in few forests outside of research plots. Consequently, the focus

³ The notable exception is a recent series of studies in Brazil which compared certified and non-certified operations across a broad geographical range and certification environments (but only post-certification; Barbosa de Lima et al. 2008). Unfortunately the direct impacts on biodiversity were outside the scope of that study.

of most studies on the biodiversity impacts of forest management activities in the tropics is on logging, RIL or otherwise. So too is the focus of more certification efforts in the tropics where, at least until the culture of forest management changes, improved logging will remain the most reasonable objective.

In temperate forests and possibly also boreal forests, there are also direct biodiversity (species) management activities (on top of forest management activities that affect biodiversity), e.g. creation of nest sites (nest boxes and other artificial nests and nesting sites), purposeful manipulation of food supply (e.g., direct feeding during winter, planting of forage and other food sources), and protection from grazing and browsing by wildlife or semi-wild cattle (e.g., hunting as a management measure). These management practices are not considered in this study.

Our analysis of 'good management practices' is not restricted to studies conducted in certified forests, as many of these practices are implemented experimentally or as a component of forest management operations that are not certified.

The various certification systems in operation around the world share many basic requirements, so we do not attempt to make a detailed comparison among them. Instead, most of the report deals with the biodiversity impacts of the forest management activities most likely to be influenced by certification.

2.3 Geographic scope

In our survey, we consider tropical, temperate and boreal forest ecosystems. Forests in the tropical, temperate and boreal biomes are subjected to a wide variety of forms of exploitation and management. To the extent possible, we focus on the most current systems in each region, which range from clear-cutting in boreal forests to single tree selection methods in the tropics.

2.4 Hypothesis

While the issue of whether certification directly benefits forest biodiversity is very relevant, it is difficult to address it with precise research questions. Both forest management and biodiversity are complex in nature, with components varying in nature, in time, and in space. Therefore, to answer the question posed by the PBL and to delimit the scope of the survey, we formulated a principal hypothesis to guide our survey and analysis. We attempted to find evidence for evaluating the hypothesis *that forest biodiversity in well-managed forest management units is higher or more intact than in otherwise similar but conventionally managed areas.*

Well-managed forest management units are defined as those that were either certified or where one or more 'good management practices' were applied at operational or experimental scales. We therefore focus on comparisons between well-managed and conventionally managed forests rather than between well-managed and undisturbed forests. We do not specifically address the question

of whether certified forests have the same value for biodiversity conservation as undisturbed forests, although we will refer to this issue when appropriate. The choice for this hypothesis reflects CBD's Principle 9 of the ecosystem approach, i.e. that '[forest] management must recognise that change is inevitable' (www.cbd.int/ecosystem). This implies that establishing the fact that forest biodiversity changes as a result of forest management, certified or not, does not discredit the value of forest certification as a management tool. In the discussion of our findings, we return to this hypothesis frequently and reformulate it for each of the management practices considered.

We did not attempt to repeat the exhaustive reviews of the (tropical) literature of logging effects by Haworth and Counsell (1999), Putz *et al.* (2000), Azevedo-Ramos *et al.* (2005), and Meijaard *et al.* (2005). This literature is summarized briefly by way of background in Chapter 4. Instead, to address the more focused question of the effects of *certified* forest management on biodiversity, we focused on studies in which different types or intensities of forest management were compared. However, in the summary of conclusions for each section, we do refer to differences in biodiversity between well-managed forests and undisturbed forests, as far as this issue was discussed in the literature reviewed.

We focused primarily on articles published in peer-reviewed journals and well-distributed edited volumes. We used the Web of Science supplemented by the Google Scholar and reference to our own extensive bibliographic collections. The main keywords used for finding relevant articles were: certification, biodiversity, species diversity, (sustainable) forest management, management practices, reduced-impact logging, riparian buffer zones, green tree retention, corridors, protected areas, and high conservation value forests (HCVF). To a limited extent and for context, we made additional use of the professional opinions of experts in the field of forest management and forest certification.

We first searched for studies describing pre- and post-certification biodiversity, whether measured directly or assumed on the basis of successful implementation of the required management practices. Next, we searched for studies in which biodiversity was compared before and after implementation of individual 'good management practices'. We primarily searched for studies conducted at the species level. In all cases we preferentially included studies that compared biodiversity within the same forest (or at least between comparable forest management units) and within the same time frame before and after logging, with a control unit that was conventionally logged. Because such studies are scarce, we also included other studies that reported clear effects of the management activity.

Even though some forest management systems and species groups have clearly been studied more intensively than others, and temperate and boreal forests are better known than tropical forests, we aimed for a mixture of studies representing as many geographical zones, forest management systems and species as possible. Where the literature on a management activity was vast (e.g., green tree retention, RIL, and the protection of riparian corridors), we only reviewed the key articles.

Tables 1 and 2 summarise the distribution of the literature used in this study over geographical zones, species groups and management practices.

2.5 Report outline and presentation of the findings

Several of the central concepts related to the main question of the study are complex and open to multiple interpretations. This caveat applies to the definitions and interpretations of biodiversity, sustainable forest management, forest certification, and their interactions. For this reason, we introduce these terms in a number of introductory chapters preceding the presentation of the findings of the survey.

The first section of the report addresses the definition of biodiversity in the context of forest certification (chapter 3), followed by an overview of what is known about the impacts of logging and other forest management activities on biodiversity (chapter 4). Next we discuss sustainable forest management (SFM, chapter 5), forest certification and the way in which certifiers assess the effects of certified forest management on biodiversity (chapter 6). After that we provide a literature review of scientific studies on the effects of certified forest management activities on biodiversity (chapter 7). We conclude by evaluating and discussing our findings (chapter 8).

The tables 1 and 2, below, give an indication of the nature of the literature on certification effects on biodiversity included in this study. Studies were about equally distributed over the three biomes included, but there were large differences between the management practices dealt with. There is a wealth of literature on reduced-impact logging, Green Tree Retention, buffer zones and corridors, but very little on certification itself, protected areas and High Conservation Value Forests. In terms of taxa considered, mammals, birds insects and plants were about equally represented, while other taxa received much less attention from scientists.

Table 1. Number of studies included in the review.

# Studies	Management Activity							Total
Zone	Certification	RIL	Riparian Buffer	GTR	Protected Area	HCVF	Corridor	Total
Tropical	1	17 ^a	1		2		4 ^c	25
Boreal	2 ^a		4	12 ^b	2			20
Temperate			9 ^a	4 ^{ac}			9 ^{ab}	22
Total	3	17	14	16	4	0	13	67

RIL: reduced-impact logging / GTR: green tree retention / HCVF: high value conservation forest

^a including 1 review / ^b including 1 simulation study / ^c including 1 study in plantation

Table 2. Distribution of studies by species group. Studies covering more than a single species group appear more than once.

# Studies	Management Activity	Plants	Insects	Mam- mals	Birds	Herpeto- fauna	Other
Tropical	Certification	1		1			1
	RIL	5	4	4	4		1
	Riparian Buffer		1		1		
	GTR						
	Protected area			1			
	HCVF						
	Corridor	1		3			
	Total	7	5	9	5	0	2
Boreal	Certification						
	RIL						
	Riparian Buffer		2		1	1	1
	GTR	5	6		1	1	
	Protected area		1				1
	HCVF						
	Corridor						
	Total	5	9	0	2	2	2
Temperate	Certification						
	RIL						
	Riparian Buffer	1		2	2	5	
	GTR	2		2	2		
	Protected area						
	HCVF						
	Corridor	3	2	3	2		
	Total	6	2	7	6	5	0
Grand total		18	16	16	13	7	4

Reviews were not included in these figures



3

What is biodiversity?

3.1 Definitions of biodiversity

The 1992 United Nations Earth Summit in Rio de Janeiro defined “biodiversity” as “the variability among living organisms from all sources, including, *inter alia*, terrestrial, marine, and other aquatic ecosystems, and the ecological complexes of which they are part: this includes diversity within species, between species and of ecosystems”. This is, in fact, the closest thing to a single legally accepted definition of biodiversity. The major certification systems (FSC, PEFC, CSA, SFI, MTCC) use the same, or very similar, definitions of biodiversity (Table 3).

Table 3. Definitions of biodiversity used in certification systems.

FSC	“Biological diversity is the variability among living organisms from all sources including, <i>inter alia</i> , terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems.” (their source: Convention on Biological Diversity)
PEFC	Not mentioned in their ‘terms and definitions’, but PEFC based its criteria on those of ITTO
ITTO	“Biological diversity or biodiversity is not just the number of species in a particular area. Rather, it is the total variety of genetic strains, species and ecosystems that are found in nature. For practical purposes biodiversity is normally subdivided into three major hierarchical categories - variation at the genetic level within a particular species; species diversity or the number and proportion of different species in a particular area; and ecosystem diversity that describes the variation in the assemblages of species and their habitats.”
CSA	Biodiversity (biological diversity) — “the variability among living organisms from all sources, including <i>inter alia</i> , terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems” (Environment Canada, Canadian Biodiversity Strategy).
SFI	“Biological diversity, biodiversity: the variety and abundance of life forms, processes, functions, and structures of plants, animals, and other living organisms, including the relative complexity of species, communities, gene pools, and ecosystems at spatial scales that range from local to regional to global.”
MTCC	“Biological diversity: the variability among living organisms from all sources including, <i>inter alia</i> , terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems.” (their source: Convention on Biological Diversity)

In reviewing the impacts of tropical forest management activities on biodiversity, Putz *et al.* (2000) disaggregated biodiversity into 5 components: landscapes, ecosystems, communities, populations/species and genes using an approach initially proposed by Noss (1990). Each of these components has structural, compositional, and functional attributes (Table 4). *Structure* refers to the physical

organization or pattern of the elements, *composition* refers to the identity and variety of the elements in each of the biodiversity components, and *function* refers to ecological and evolutionary processes acting among the elements. Logging and all other forest management activities have impacts on each component and each attribute of biodiversity, although some are more sensitive than others.

The definition of biodiversity and the disaggregation mentioned above are suitably comprehensive, but due mostly to lack of information, this report focuses primarily at the species and community level as outlined in section 2.1. This limitation is consistent with the species-level concerns of most environmentalists, researchers, foresters, policy-makers, and other actors in forest policy and management.

Table 4. Components and attributes of biodiversity (Putz et al. 2000).

Components	Attributes		
	Structure	Composition	Function
Landscape	Size and spatial distribution of habitat patches (e.g., seral stage diversity and area); physiognomy; perimeter-area relations; patch juxtaposition and connectivity; fragmentation.	Identity, distribution, and proportion of habitat types and multi-habitat landscape types; collective pattern of species distributions.	Habitat patch persistence and turnover rates; energy flow rates; disturbance processes (e.g., extent, frequency, and intensity of fires); human land use trends; erosion rates; geomorphic and hydrologic processes.
Ecosystem	Soil (substrate) characteristics; vegetation biomass; basal area and vertical complexity; density and distribution of snags and fallen logs.	Biogeochemical stocks; lifeform proportions.	Biogeochemical and hydrological-cycling; energy flux; productivity; flows of species between patches; local climate impacts.
Community	Foliage density and layering; canopy openness and gap proportions; trophic and food web structures.	Relative abundance of species and guilds; richness and diversity indices; proportions of endemic, exotic, threatened, and endangered species; proportions of specialists vs. generalists.	Patch dynamics and other successional processes; colonisation and extinction rates; pollination, herbivory, parasitism, seed dispersal and predation rates; phenology.
Species/ Population	Sex and age/size ratios; range and dispersion.	Species abundance distributions, biomass, or density; frequency; importance or cover value.	Demographic processes (e.g., survivorship, fertility, recruitment, and dispersal); growth rates; phenology..
Genetic	Effective population size; depression; heterozygosity; polymorphisms; generation overlap; heritability.	Allelic diversity; presence of rare alleles; frequency of deleterious alleles.	Gene flow; inbreeding depression; rates of outbreeding, genetic drift and mutation; selection intensity; dysgenic selection.

3.2 Why is it important to conserve biodiversity?

Biodiversity forms the foundation of the vast array of ecosystem services (if one follows the definition of biodiversity of Table 4, these ecosystem services are even included in the definition of biodiversity) that contribute to human well-being (MEA 2005). But how important is ecosystem functioning for society and why is it important to preserve biodiversity?

Plants, animals and microbes ensure the production of food, clean water, fuels and medicine. They capture and convert energy into a variety of materials, they process waste, drive global biochemical cycles, and regulate the world's climate. In general, they affect the strength and capacity of ecosystems to provide essential goods and services necessary for the well-being and prosperity of human populations in both developed and developing countries (Martens *et al.* 2003; MEA 2005).

Biodiversity loss is important in its own right because biodiversity has cultural values, because many people ascribe intrinsic values to biodiversity, and because it represents unexplored options for the future. People from all walks of life value biodiversity for spiritual, aesthetic, recreational, and other cultural reasons. Species extinction at the global level is also of particular significance, since irreversible losses of species represent losses in the constitutive elements of well-being. Population extirpation is particularly important at national and local levels insofar as most ecosystem services are delivered at the local and regional level and are strongly dependent on the types and relative abundances of species (MEA 2005).

Biodiversity loss is closely associated with the rapid growth in the human population as well as to increased per capita rates of consumption. The most important direct drivers of biodiversity loss and change in ecosystem services are habitat loss and land-use change, physical modification of rivers, water withdrawal from rivers, loss of coral reefs, and damage to sea floors due to trawling, in addition to climate change, invasive alien species, overexploitation of species and pollution (MEA 2005). Biodiversity conservation is imperative for diminishing detrimental effects on ecosystem functioning.

3.3 Measuring biodiversity

While it is increasingly clear that biodiversity contributes substantially to ensuring ecosystem functioning and human well-being, its measurement remains challenging. One of the difficulties in measuring biodiversity is that because of the huge variation in species, especially at lower hierarchical levels, and the complex relations between species and ecosystem functioning, our understanding of biodiversity is relatively limited. For example, in a comparatively well studied forest in the Pacific Northwest, researchers estimated that 400 of the 1100 catalogued species were rare and poorly known (Rafael and Molina 2007). Furthermore, for all ecosystems it is difficult to determine which species can be lost without substantial detriment, and which losses may cause the system to disappear. This challenge is even greater taking into account the large geographical variety in

biodiversity, which makes generalizations difficult (Martens *et al.* 2003). Apart from these functional aspects it is even hard to decide which metric best 'captures' the biodiversity of a given area. Should this be number of species (=species richness), should it reflect species identity, or should the relative abundances of different species be considered?

The simplest and seemingly most straightforward measure of biodiversity is the number of species present in a specified area. Unfortunately, species richness alone does not fully capture biodiversity. For instance, it does not take into account which particular species are present or how many individuals there are per species. In simple tallies of species, widespread and common taxa are weighted the same as those that are rare or endemic, those that play important ecological roles, and those that are otherwise noteworthy. Furthermore, by including no information about the numbers of individuals per species, taxa likely on the brink of extirpation due to small and non-viable population sizes are not distinguished from the most common taxa. Species richness values alone also do not distinguish between species that fulfill critical roles in the ecosystem, for instance pollinators, and those that play less important roles. Neither does it differentiate between native and non-native species, such as exotic, introduced, or invasive species that disrupt ecosystem services (MEA 2005). Ghazoul and Hellier (2000) suggested that species richness alone may not be a good indicator of the recovery of forest biodiversity, given that there are no consistent patterns among the studies they reviewed. Ideally, biodiversity measures should recognize different species values, but so far no such approach has proven to be fully satisfactory or been accepted.

This conclusion was also drawn in a comparative study on the performance of a number of metrics for monitoring biodiversity change. Lamb *et al.* (2009) compared four major approaches for translating complex monitoring data into easily communicated summary statistics: (1) traditional diversity indices such as species richness and Simpson's index (2) species intactness indices based on occurrence, (3) species intactness indices based on abundance, and (4) multivariate community indices. Of 13 metrics compared, Buckland's arithmetic mean index, including aspects of species' observed and reference abundances, outperformed others including species number in detecting changes in biodiversity.

For specific purposes, such as for modeling different scenarios of biodiversity loss, the dilemma of selecting an appropriate biodiversity metric must be addressed. The Netherlands Environmental Assessment Agency (PBL) developed a measure of biodiversity which is used in advising national and international policy-makers and the CBD (CBD/MNP 2007). The Mean Species Abundance (MSA) is a measure of the species abundance (i.e., the number of individuals per species) in a disturbed area relative to the abundance in the natural state of that area. Invasive and exotic species are not taken into consideration. Data to calculate this measure are taken mostly from experimental studies reporting on species abundances before and after disturbance, reported in peer-reviewed articles (Bouwman *et al.* 2006).

Like all metrics, the MSA index is a simplification of the over-arching concept of biodiversity insofar as it regards diversity only at the species level. Also, because all species are assigned equal value in MSA calculations, the index suffers from some of the same deficiencies as outlined above for measures of species richness. Although introduced species are disregarded, unique, threatened, or otherwise “special” species are not treated differentially in the index. MSA is calculated based on available data and is thus prone to bias because, for example, more research is done on plants than on microbes. These apparent (and partly unavoidable) limitations notwithstanding, the MSA is one of the indicators being used worldwide to study the biodiversity effects of several factors land use, population pressure etc. (CBD/MNP 2007). Nevertheless, in the review in this report as in many other applications, it is challenging to use the MSA approach without a clear definition of original forest species and a good idea of reference (pre-logging) densities.





4

Impacts of logging on biodiversity

4.1 Primary and secondary impacts

Forests differ in biodiversity, in their capacity to support different intensities of silvicultural use, in pressures for conversion to non-forest use, and in the abilities of the relevant institutions to regulate their management (Putz *et al.* 2000). All forest management interventions, no matter how reputedly 'sustainable', have impacts on biodiversity.

Logging impacts on biodiversity of primary, undisturbed forests have been the subject of many studies. In this chapter, we briefly review this literature in order to provide the context for the discussion in Chapter 7 of the impact of introducing certified forest management.

Fundamentally, we distinguish between the primary and secondary impacts of forest management activities. "Primary" impacts are the direct effects of road building, tree felling, log yarding, and log hauling. Such impacts include soil damage that results in compaction and erosion, damage to the residual stand in selectively logged forests, changes in forest microclimate, changes in food availability, and loss of habitat (e.g., trees used for nesting and roosting). "Secondary" impacts, which can be more serious, are mostly related to the improved access provided by logging roads. This access, if uncontrolled, can facilitate spontaneous forest colonization and conversion as well as increased rates of forest fire along with wildlife and timber poaching. Forest certification recognizes these different impacts and assesses the extent to which both the primary and secondary impacts of forest management are mitigated by managers. Unfortunately, these impacts are not always adequately distinguished in the literature. Ultimately, if forest certification successfully reduces the direct biodiversity impacts of forest management but fails to affect the secondary impacts, it cannot be judged successful in regard to biodiversity conservation.

Logging can be carried out in ways that minimize the deleterious environmental impacts, or can be massively destructive. "Good logging" may seem like an oxymoron to many people, but there are actually many ways by which conscientious and informed loggers can diminish the negative impacts. In fact, where properly carried out, logging is a silvicultural treatment, not just forest exploitation. For example, by opening the canopy, logging can be used to promote the establishment and growth of light-demanding commercial species. In contrast, where logging does not result in the conditions appropriate for regeneration of the harvested species, it is essentially a timber mining activity, not a component of forest management.

4.2 Effects of logging on biodiversity

Logging and other timber stand management activities directly and indirectly affect all five components of biodiversity (Putz *et al.* 2000): landscapes, ecosystems, communities, species, and genes (Table 4). Discussions of even the primary impacts of forest activities on biodiversity are complicated due to the wide range of possible forest management activities and the even wider range of forests and responses to these interventions. Even at the species level the direct impacts are difficult to describe because species respond in distinct ways to logging and associated activities (see below in section 4.2.4).

4.2.1 Landscape

Although discussions of the landscape-level impacts for forest management activities are better informed when the landscape setting is specified, authors often fail to contextualize their research sites. For example, forest practices in frontier areas might be expected to be very different from those used in areas where only habitat fragments remain. Chomitz (2007) describe this gradient in forest settings from the densely settled mosaic lands where laws are typically enforced to the frontier forests where populations are scarce, transport costs are high, and even land tenure security is limited.

At the landscape level, logging and other forest management activities affect biodiversity by changing land forms, forest cover, and ecosystem types across large geographic areas. As the intensity and spatial extent of interventions increases, habitat patches increasingly change in size, identity, spatial distribution, and connectivity, all of which affect species distribution patterns. In the more remote areas, due partially to the high transport costs for low value timber, the more severe impacts often result from the indirect consequences of logging that result from increased access such as over-hunting, spontaneous colonization and altered fire regimes.

While deforestation results in more intensive forest fragmentation than logging, wide logging roads can impede species movements and isolate sub-populations, while creating access corridors for the invasion by secondary and non-forest species. For understory species that will not cross canopy openings, the degree of population fragmentation varies with whether logging is dispersed over large areas or concentrated in small areas. Even though fires play a substantial role in the ecology of some forest ecosystems, in forests that are not naturally fire-maintained, by opening the canopy and thereby increasing the rate of forest drying, selective logging increases forest flammability and the intensities of fires that do occur. The synergistic effects of logging and fire can lead to forest replacement by savanna or scrub, with all of the attendant losses in forest biodiversity (e.g., Nepstad *et al.* 1999).

4.2.2 *Ecosystem*

Forest management activities have a multitude of impacts at the ecosystem level, which vary with the intensity of logging and other silvicultural treatments and the care with which these operations are carried out. Because ecosystems vary in their responses to even the same intervention, the diversity of ecosystem responses is substantial. Fundamentally, the effects are on spatial heterogeneity, carbon stocks and fluxes, and soil characteristics. The extent of the impacts varies with logging intensity, the size of the trees extracted, the yarding methods used, and the care with which the operations are carried out.

Biomass losses result not only from timber extraction itself, but also from damage to the trees in the residual stand and subsequent silvicultural treatments. By opening the canopy, and especially by creating large canopy gaps, these treatments increase the risks of uncontrolled fires and increase the intensities of fires that do occur. If managed stands are colonized by lianas and other low-biomass forest weeds (e.g., gingers and pioneer trees with low density wood), forest carbon stocks can be depressed for decades. If soils are compacted or otherwise rendered unproductive during harvesting operations, reductions in the carbon storage capacity of the stand can be more permanent. Soil compaction as a result of ground-based yarding also reduces the water infiltration rates and water holding capacity of soils, which leads to increased surface run-off. Impoundments caused by road building, bridge collapse, undersized and inappropriately set culverts, and other poor harvesting practices can also have long-term consequences for ecosystem functions and biodiversity. Deposition of sediments in streams during and after road construction or due to erosion from skid trails and log landings also affect the characteristics of water courses. Thus, flow regimes of natural streams can be greatly influenced by logging. These changes have grave impacts on ecosystem functioning and thus on (aquatic) species.

4.2.3 *Community*

Logging, especially when followed by silvicultural treatments to promote the establishment and growth of commercial timber species, changes the proportions of species and successional stages in forests. For instance, logging could affect the abundance of light demanding vs. shade tolerant species, or wind vs. animal dispersed seeds. If logging is followed by stand refinement treatment releasing future crop trees from competition, rare, threatened or endangered species without commercial value may be lost from the forest. In contrast, where light-demanding tree species are harvested, unless the canopy is sufficiently opened, their populations are likely to diminish, which will have both environmental and economic consequences.

4.2.4 *Species*

The most obvious impacts of logging are on the abundance and age or size class distributions of species. Depending on the intensity of logging and the care with which it is carried out, the reproduction, growth and survival of a great number of species can be adversely affected.

If not managed sustainably, the commercial species population will be depleted and causes a direct loss in biodiversity. After logging, many formerly shaded areas in the forest interior become drier, warmer, brighter, and more easily exploited by some predators which will affect the specialist species, and in some cases also generalist species. Population sizes and structures are also affected by fires that often follow uncontrolled logging.

The impacts of logging persist for many years after it is completed, particularly if badly done (Kariuki *et al.* 2006; Putz *et al.* 2000 for references). Even though logging may have no detectable negative short-term effect on many species, the indirect and long-term effects might be quite severe. One cause of continued degradation of badly managed stands is the continued high mortality rates of trees in the residual stand, often due to apparently slight but eventually fatal logging-related damage. Also, the proliferation of weeds and changes in the services provided by seed dispersal agents and pollinators due to population size reduction and fragmentation can have persistent effects. Animal species composition also changes in response to logging, as well to the direct effects such as canopy opening, and indirect effects, such as increased hunting pressure, fires, and forest conversion. Disturbance-adapted species that are not native to the area often proliferate in logged-over forests and affect the resident flora and fauna.

Although logging has a variety of deleterious effects at the species level, the changes in species composition in response to forest activities are by no means consistent across or even within taxa. This species and stand-specificity can partly be attributed to differences in logging intensities and in differences in post-logging population monitoring.

From scientific studies it is known that plants and animals show highly variable, unpredictable responses to logging but we attempt to make some generalizations. For example, unless appropriate silvicultural treatments are applied, populations of the harvested tree species are typically depleted, especially after a few logging cycles (Grogan *et al.* 2008; ter Steege 2003). Often the result of this depletion is that light-demanding or “weedy” species appear while shade-demanding species with high density wood decline in numbers (Kariuki *et al.* 2006; Meijaard *et al.* 2005). In the case of animals, chimpanzee populations have been reported to increase, decrease, and not respond at all to logging. In contrast, terrestrial and bark-gleaning insectivorous birds consistently show negative impacts of logging. Frugivorous canopy birds reportedly show slightly negative, neutral or even positive population-level responses to selective logging (see also Ghazoul and Hellier 2000; Meijaard *et al.* 2005; Thiollay 1997). Large and slow-reproducing animals are sometimes severely affected not only because of the direct effects of logging but more so because of the increased hunting pressure after forests are made more accessible by the construction of logging roads. Insectivorous and carnivorous bats and hornbills usually decline in numbers after logging whereas populations of other species such as browsing and grazing ungulates often increase due to the proliferation of grasses and shrubs after canopy opening (Bennett 2003). In general, the more ecologically

specialized a species is, the higher the probability of a decline in population size or even local extirpation after logging (Meijaard *et al.* 2005).

Table 5. Effects (+ positively; - negatively; 0 no effect) of selective logging in tropical forests on avian communities and feeding guilds in the Neotropics, the African tropics and the Oriental tropics. This demonstrates the variation in responses to logging, even for functionally similar species. Taken from (De longh and van Weerd 2005).

Guild	Neotropics	Africa	Oriental
Community: diversity/ abundance	4 (0) 7 (-)	7 (+) 4 (0) 2 (-)	2 (+) 4 (0) 8 (-)
Carnivores	2 (+) 1 (-)	--	1 (+)
Nectarivores	3 (+) 2 (0)	--	3 (+)
Frugivores	6 (+) 1 (-)	3 (+) 2 (-)	4 (+) 4 (0) 4 (-)
Insectivores	7 (-)	2 (+) 3 (-)	5 (-) 5 (0)
Total number of studies	15	9	14

4.2.5 Genes

By reducing the effective population size the genetic component of biodiversity is affected. For instance when a large proportion of healthy reproductive adults is being harvested, especially when this is coupled with losses of pollinators and seed dispersal agents, the genetic variation may change. However, the techniques required for assessing the genetic structure of populations are sophisticated and expensive, and most concerns about genetic issues in relation to logging are still inferences from theory rather than demonstrated in practice (cf. Putz *et al.* 2000).





5

Sustainable Forest Management

Existing parks and protected areas are cornerstones of biodiversity conservation but on their own inadequate to assure the continued existence of a vast proportion of (tropical) forest biodiversity. As a result, priority must be given to ensuring that the greatest possible amount of biodiversity is conserved outside protected areas in production forests, which occupy large areas. From a biodiversity maintenance perspective, sustainable natural forest management is preferable to virtually all land-use practices other than complete protection (Putz *et al.* 2000).

5.1 What is sustainable forest management?

As societal concerns about the fates of forests increase and human demands on forests change, so are the ways in which they are silviculturally treated. A few decades back the primary component of sustainability on which many people focused was sustained timber yields. However, due to the high rates of deforestation and loss of biodiversity, this focus has recently broadened to encompass a wide range of factors constituting sustainable forest management (SFM).

The definition of SFM developed by the Ministerial Conference on the Protection of Forests in Europe (MCPFE) and adopted by the Food and Agriculture Organization (FAO) is 'the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems'. In simpler terms it means attaining a balance between society's increasing demands for forests and forest products and the preservation of forests health and biodiversity. It should be pointed out that the desired outcome of SFM is not fixed. Forests and societies are in constant flux and thus the idea of what constitutes a sustainably managed forest changes over time. Moreover, stakeholders involved in SFM generally have different visions of what sustainable management entails. The preservation of species, particularly red-listed, rare or endemic species is the principle aim of conservation biologists whereas the maintenance of a sustainable resource is of central concern to a forest manager. Communities living in and around production forests, employees and other stakeholders have yet different expectations from forest management. However, in general, multi-stakeholder agreement on SFM has been reached for different geographical areas and purposes, and has been codified in terms of principles criteria, and, sometimes, indicators.

At the international governmental level, these sets of principles, criteria and indicators were a response to the 1992 United Nations Conference on Environment

and Development (UNCED) in Rio de Janeiro. Two of the more advanced initiatives are those of the Working Group on Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests (also called the Montreal Process), the MCPFE for European forests. Other organizations also developed sets, notably the ITTO and CIFOR for tropical forests. In 2003, about 150 countries were engaged in one or more international processes to develop national level criteria and indicators for SFM (Rametsteiner and Simula 2003).

The purpose of Criteria and Indicators for SFM is to provide countries and individual forest managers with a tool for assessing changes and trends in forest conditions and management systems. The indicators provide the information needed to monitor change, both in the forest (outcome indicators) and in the environmental and forest management systems used (input and process indicators) and measure progress towards SFM. The general recommendation is that indicators should be specific to each nation or management unit in order to assess sustainability of forest management and they should be reviewed and refined periodically. Forest certification has another purpose (namely to provide assurance to consumers that timber has been 'sustainably' produced) but makes use of comparable sets of Principles, Criteria & Indicators to assess the way in which timber operations are managed (see section 6.1).

5.2 Logging and sustainable forest management

When properly planned and conducted, logging can be an integral component of SFM. There is a wide range of logging intensities, yarding methods, and accompanying forest management practices to which forests are subjected. The substantial number of studies conducted on the impacts of logging have virtually all concluded that soil impacts and damage to the residual forest increase with increasing logging intensity (Haworth and Counsell 1999, Putz et al. 2000, Azevedo-Ramos et al. 2005, and Meijaard et al. 2005), but vary strongly with the extraction method used. Once logging and extraction are planned and carried out with care so that as little as possible unnecessary damage is done to the forest, one speaks of reduced-impact logging (RIL). Rules regarding logging and extraction are set by national governments (for instance, in codes of practice) and are expected to be applied by all forest managers.

While applying RIL is generally considered to reduce negative effects of logging on biodiversity, this is not its prime objective. Meijaard *et al.* (2005) proposed some measures in addition to RIL techniques to reduce negative effects on fauna. They suggest that:

- Forest managers should make maps of the areas with rare, unusual, and sensitive habitats and species;
- plans to conserve these habitats and species should be made;
- at least 10 % of the commercial forest area should be protected as refuges and as recolonisation sources, and be connected by corridors;

- it is important to maintain habitat complexity and diversity by, for instance, retaining hollow trees and avoiding disturbances in riparian zones;
- keystone resources such as figs, should be conserved;
- post-logging operations should be designed to promote species recovery from the disturbance. Appropriate treatments include closing logging roads (even temporarily) and replanting trees. Hunting and wildfires should be prevented;
- invasive and exotic species should be removed, and certainly not planted, and traffic should be controlled to prevent illegal trade in forest products.

Note

For sustaining timber yields, low impact logging is not always the most suitable practice (Fredericksen and Putz 2003). While RIL techniques that protect soils and hydrological functions are always appropriate where biodiversity conservation and forest production are concerned (Putz *et al.* 2008), sustaining the yields of some commercially valuable species, particularly those that are light-demanding, requires more forest disturbance than is typically the goal of RIL. The most familiar example of this phenomenon is mahogany (*Swietenia macrophylla*), which in many forests regenerates best after hurricane-like disturbances followed by competition-reducing fires (Fredericksen and Putz 2003). A similar pattern can be seen in some Cameroonian forests where most upper canopy trees prefer shifting cultivation fields for recruitment (van Gernerden *et al.* 2003). In contrast, in forests where there is abundant “advanced regeneration” (i.e., seedlings, saplings, and pole-sized trees) of commercial species, such as in the dipterocarp forests of Southeast Asia, protecting the residual forest is critical and RIL is tantamount to sustained yield forest management (Meijaard *et al.* 2005).



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Forest certification

Parallel with the development of criteria for evaluating progress towards SFM, the concept of voluntary timber certification emerged in response to growing environmental awareness and consumer demand for more responsible business (Upton and Bass 1996; Viana *et al.* 1996; Vogt *et al.* 2000). Forest certification is an initiative of conservation NGOs and the private sector, who were dissatisfied with the rate of progress in formal discussions on promoting SFM. It should assist consumers in verifying that timber in the market comes from a well managed forest, with an eye for economical, social and environmental issues.

6.1 Forest certification standards

Several certification systems emerged throughout the world. The first system to become functional was managed by the Forest Stewardship Council (FSC), established in 1993, followed by the Programme for the Endorsement of Forest Certification schemes (at that time called Pan European Forest Certification, PEFC) in 1999. FSC uses its own principles and criteria whereas PEFC uses different standards in different areas, mainly based on the MCPFE and ITTO. In addition to these international schemes, several national standards have been developed, of which the largest are mentioned. The Malaysian Timber Certification Council (MTCC) developed a set of criteria and indicators based on those of ITTO. Lembaga Ekolabel Indonesia (LEI) operates a standard for Indonesia, which is said to be comparable and compatible with that of FSC (Hinrichs and Prasetyo 2002). The Sustainable Forest Initiative (SFI) and the Canadian Standards Association (CSA) are the main systems in North-America. Both the SFI and the CSA are endorsed by the PEFC. FSC has endorsed national FSC forest management standards ('national initiatives') in more than 46 countries, based on the generic FSC Principles and Criteria.

Forest certification aims to promote sustainable forest practices through independent evaluation of forest management against a number of requirements that are generally described as "Principles and Criteria" (Table 6). *Principles* are essential rules or elements whereas *Criteria* are the means of judging whether or not a principle has been fulfilled. In the evaluation process in certified forests (i.e., during audits) these requirements translate into a number of indicators to ascertain whether management plans and practices meet the requirements. An *Indicator* is any variable or component of the forest ecosystem that is used to infer the status of a particular criterion. Indicators can potentially be used to measure the success or failure of management practices to sustain biodiversity (Lindenmayer *et al.* 2000). Indicators are measured or assessed in the field and form the basis for judging the Criteria and through these, adherence to the Principles.

Table 6. Definitions of Principles, Criteria and Indicators.

Principle	A principle is a fundamental law or rule, serving as a basis for reasoning and action. Principles have the character of an objective or attitude of society concerning the function of the forest ecosystem or concerning a relevant aspect of the social system that interacts with the ecosystem. Principles are explicit elements of a goal e.g., sustainable forest management or well managed forests.
Criterion	A criterion is a state or aspect of the dynamic process of the forest ecosystem, or a state of the interacting social system, which should be in place as a result of adherence to a principle of sustainable forest management (or well managed forest). The way criteria are formulated should give rise to a verdict on the degree of compliance in an actual situation.
Indicator	An indicator is a quantitative or qualitative parameter which can be assessed in relation to a criterion. It describes the features of the ecosystem or the related social system in an objectively verifiable and unambiguous way, or it describes elements of prevailing policy and management conditions and human driven processes indicative for the state of the eco- and social system.

Source: Lammerts van Bueren and Blom (1996)

The development of standards is the first step in forest certification. Producers who wish to demonstrate that they meet the standards can voluntarily request certification of their forest operation. This is accomplished by third-party certification, by a certifying organization that is accredited by the standard setting body. If a forestry company passes the audit, often after being required to adapt its management practices, it is said to be 'certified' and eligible to use a logo to distinguish its wood products in the market. After the initial certification some systems require minor annual audits to ensure that there have been no major changes in management that would contradict the forestry standards, and to ensure that any required corrections to management have been implemented. After a specific period (5 years for FSC) another comprehensive audit is required (Gullison 2003).

The variety of certification schemes, the existence of "part-way to FSC" certification systems (such as TFF's RIL certificate, TFF 2008) and various legality standards that partly address the issue of sustainable forest management complicate an evaluation of 'the' biodiversity effects of certification.

The complexity of setting standards for certification is enormous. Not only are forests highly diverse around the globe, varying from dry shrub land in Australia to tropical rain forest in Brazil and Congo to boreal forests in Russia, but substantial variation is also found within geographical regions. Furthermore, management of these areas varies greatly depending on ecosystem characteristics and management goals. Also in need of consideration is the large range of stakeholders, varying from private forest owners to environmental NGOs to indigenous groups and multinational corporations; in each case, numerous stakeholders want to be recognized and participate in determining how forests should be managed. Consequently certification caters for many different stakeholders and their respective interests. As a result, forest certification standards vary considerably, reflecting the diversity of stakeholder views and local conditions (Gullison 2003; Rametsteiner and Simula 2003). This variation translates into a high local specificity of the Indicators used to assess forest management, although (within a certification scheme) the Principles

and, to a lesser extent, the Criteria of SFM are more or less universal. The variation in standards renders cross-country and cross-scheme comparison of standards and audit results an imprecise exercise. The same holds logically true for 'the' impacts of 'certification' on biodiversity.

6.2 Is certified forest management sustainable?

The existence of certified forest operations – which are assessed against standards claimed to represent sustainable forest management – raises the issue whether it can be assumed that certified forests are sustainably managed. There is a difference in what many consumers and some policy-makers believe that a timber certificate represents and what timber certification agencies actually assess and what they can legitimately claim. Many may believe that certified timber is derived from forests that are being managed sustainably insofar as the full complement of biodiversity and ecological functions of the forest are being conserved. In reality, these forests satisfy nothing more and nothing less than the criteria of a certification standard. One may question whether these standards capture 'sustainability':

- Each standard is a 'social contract' and a compromise between different, sometimes incompatible interests;
- each standard contains tradeoffs between what is desirable and what is practically feasible in forest management;
- there may be differences for the same forest between certification schemes⁴, suggesting that there are, apparently, different interpretations of sustainability;
- there are limitations to both certified forest management in achieving, and to auditing in assessing, all dimensions of the standard. "No guidelines indicate which criteria and indicators must be enforced, or to what degree, for certification to be conferred by third-party assessors" (Schulze *et al.* 2008);
- there are limitations to the knowledge about species responses to forest management, and therefore no objective guidelines can be established for evaluating compliance with criteria and indicators regarding these species (Schulze *et al.* 2008).

All these considerations complicate assertions of the sustainability of forest management in a given forest. As a result, it is hard to be certain of what a certificate really represents, and it is hard to defend 'absolute' sustainability of any of them in all senses of the word. PEFC, SFI, CSA, and MTCC contribute to this misunderstanding

⁴ In an analysis of Forestry Certification Programs in Canada (FSC, CSA and SFI), it was concluded that the standards differed greatly in their requirements on how sustainable forestry should be achieved (EEM 2007). For instance, much less cutting is allowed in an FSC-certified forest (Annual Allowable Cut of 0.43 m³/ha.yr-1), compared to a CSA-certified forest (0.75 m³/ha.yr-1) and, particularly, SFI-certified forests (1.27 m³/ha.yr-1). Other examples include the conversion of natural forest into plantations, which is prevented by FSC but not by the others, and the application of the precautionary approach to the management of HCVF which is required by FSC but not by CSA and SFI (EEM 2007). The Malaysian Criteria and Indicators accept logging on slopes up to 45 degrees whereas most others accept harvesting to slopes < 20 degrees (Putz *et al.* 2008), despite the availability of substantial evidence showing that the deleterious ecosystem impacts of logging, especially with ground-based extraction, increase with terrain slope.

by claiming that they certify “sustainably managed forests”. The FSC, in contrast, is careful to state that its timber comes from “well-managed forests”.

There are a number of different ways in which the “sustainable” component of SFM can be considered but most definitions include ecological, social, and economic factors as the pillars of sustainability. The diversification of requisites for sustainability in regards to forest management reflects the increasing influence of non-foresters on the forestry agenda and an increasing understanding of the importance of forests to society and the environment. In part it can be seen as a response to the former focus of many foresters on sustaining timber yields, often at substantial costs to other environmental and social factors. In a provocatively titled paper (“What should forests sustain? Eight answers”), Gale and Cordray (1991) propose a more elaborate definition of “sustainability” that still includes consideration of sustained yield (“dominant product sustainability” in their parlance), but also considers sustainability from community, human benefit, “global village” self-sufficiency, ecosystem type, ecosystem insurance, and ecosystem-centered perspectives.

The difference between ‘certified’ and ‘sustainable’, in this case in its more limited interpretation of sustained yield, is illustrated by the case of cutting cycles in seventeen forest management units in the Brazilian Amazon. The projected cutting cycles varied between 25-30 years, however at the actual harvest intensity most companies would exhaust their production area within 4-20 years, with just three companies able to adhere to the projected cutting cycle without increasing land (production) area or halting operations for a number of years (Schulze *et al.* 2008).

In this report we will not follow the familiar convention of using the term ‘sustainable forest management’ loosely to mean well-managed. Instead, we will continue to refer to SFM as the ultimate goal, just as we hope for justice and peace. A further justification for this careful use of terminology is that the charge given to us from PBL was to evaluate the biodiversity impacts of certified forestry; if ‘certified’ and ‘sustainable’ were equivalent, then there would appear to be no need for this study.

6.3 Biodiversity indicators in forest certification standards

For a variety of reasons, direct biodiversity indicators measuring species richness or abundance are rarely used in certification systems and as a result, biodiversity is rarely determined directly during certification audits. These indicators are costly and time-consuming to measure, and high variation hampers interpretation. Instead, management process indicators are measured which attempt to ensure that inherent taxonomic, structural, and landscape complexities characteristic of forest ecosystems are maintained and in so doing contribute to the conservation of biodiversity (Lindenmayer *et al.* 2000). Guynn *et al.* (2004) reviewed biodiversity-related indicators used by several certification systems (FSC, PEFC, SFI, CSA) and identified two types:

- Those designed to reflect the type and status of management processes and
- those that provide actual measures of forest components to allow comparisons against desired outcomes and standards.

The first type of indicators relate to the use of mapping systems, the existence of biodiversity goals in harvest plans, compliance with laws, etc. The second kind, outcome-indicators, are only vaguely described in most systems and are most often intended to reflect the presence of selected structural features or forest processes. Table 7 lists metrics as they appear in a variety of certification standards.

Forests, forest biodiversity and local perceptions of sustainable forest management vary throughout the world, calling for locally or nationally adjusted certification standards. This is shown in Table 8 for the example of the FSC standard, which are highly standardized at the Principles and Criteria level. The example illustrates how indicators are specified at a generic (world-wide) and national (Australian) level, for a criterion of high relevance to biodiversity.

Table 7. Metrics of certification standards applying to biodiversity (Guynn et al., 2004 - see that reference for details about their use in certification systems).

Stand-level, outcome-oriented metrics	Landscape-level, outcome-oriented metrics
Age, size, and species diversity of trees	Ecological function, cycles, and productivity
Dead wood	Ecological reserves or high conservation value forests
Excessive herbivory by deer	Examples of existing ecosystems
Disturbance by biotic and abiotic agents	Exotic species
Herbicide, pesticide, and/or biological control	Fire, prescribed burning
Mixed species stands	Fragmentation
Presence or distribution of hardwoods and broadleaved trees	Mature or old-growth stands
Road management and habitat inputs	Natural regeneration, deforestation, plantations
Rotation length	Rare or unique physical environments
Soils characteristics, function, nutrient capital	Restoration of forest types, refugia
Understorey species diversity	Seed source, genetically modified organisms
Vertical and horizontal stand structure	Water course or wetlands

Table 8. Translation of Principles and Criteria into regional Indicators, illustrated by the example of FSC Criterion 6.3 and its elaboration in SmartWood generic⁵ and Rainforest Alliance/SmartWood Australian standards⁶. This criterion is strongly related to forest biodiversity.

Principle 6. Forest management shall conserve biological diversity and its associated values, water resources, soils, and unique and fragile ecosystems and landscapes, and, by so doing, maintain the ecological functions and the integrity of the forest.

Criterion 6.3 Ecological functions and values shall be maintained intact, enhanced, or restored, including:

- a. Forest regeneration and succession;**
- b. Genetic, species, and ecosystem diversity;**
- c. Natural cycles that affect the productivity of the forest ecosystem.**

Rainforest Alliance SmartWood generic standard Indicators (version August 2005)	Rainforest Alliance/SmartWood Standard for Assessing Forest Management in Australia Indicators (version May 2008)
6.3.1 Silviculture and/or other management systems shall be appropriate to the ecology of the forest and other resources (e.g. soils, hydrology).	6.3.1 The forest manager shall have site-specific data or published analyses of local forest ecosystems that provide information on the FMU with regards to: <ul style="list-style-type: none"> • regeneration and succession • genetic, species and ecosystem diversity • natural cycles that affect productivity
6.3.2 Ecological and silviculture rationale behind management prescriptions shall be well-documented, based on site-specific forest data or published analyses of local forest ecology (e.g. regeneration and succession) or silviculture.	6.3.2 Forest management systems shall maintain, enhance or restore ecological functions and values of the FMU based on the data in 6.3.1. Management systems shall include: <ul style="list-style-type: none"> • silvicultural and other management practices which are appropriate for forest ecosystem function, structure, diversity and succession • where appropriate, a program for the restoration of degraded sites • natural regeneration, unless data shows that enrichment planting or artificial reforestation will enhance or restore genetic, species or ecosystem diversity
6.3.3 Management prescriptions should maintain, enhance or restore forest composition (i.e. species numbers and diversity) and structure.	6.3.3 Ecological and silvicultural rationale behind management systems shall be well-documented, based on site-specific forest data or published analyses of local forest ecology (e.g. regeneration and succession) or silviculture.
6.3.4 Management is designed to ensure that the full complement of tree species regenerates successfully in the forest area over the duration of the rotation.	6.3.4 Management prescriptions should maintain, enhance or restore forest composition (i.e. species numbers and diversity) and structure.
6.3.5 Standing and fallen dead wood habitats should be retained, based on local best management practice or documented research.	6.3.5 Management is designed to ensure that the full complement of tree species regenerates successfully in the forest area over the duration of the rotation. Where artificial regeneration is planned, environmental impact has been assessed.

5 www.rainforest-alliance.org/forestry/documents/generic_standards.doc

6 www.rainforest-alliance.org/forestry/documents/smartwoodaustraliainterimstandardsmay08.pdf

Table 8 (continued).

Principle 6. Forest management shall conserve biological diversity and its associated values, water resources, soils, and unique and fragile ecosystems and landscapes, and, by so doing, maintain the ecological functions and the integrity of the forest.

Criterion 6.3 Ecological functions and values shall be maintained intact, enhanced, or restored, including:

- a. Forest regeneration and succession;
- b. Genetic, species, and ecosystem diversity;
- c. Natural cycles that affect the productivity of the forest ecosystem.

Rainforest Alliance SmartWood generic standard Indicators (version August 2005)

Rainforest Alliance/SmartWood Standard for Assessing Forest Management in Australia Indicators (version May 2008)

- 6.3.7 The forest manager shall ensure that regeneration of native forests and establishment of plantations is effective and timely. Species composition and the density of the regeneration of native forests and the stocking rate of plantations shall be assessed and remedial action taken where necessary to ensure effective regeneration and establishment (AZ 4.4.4).
- 6.3.8 Topsoil displacement from the planting area is not permitted.
- 6.3.9 Measures are taken to reduce or eliminate impacts on aquatic resources.
- 6.3.10 In the management of native forests, FME shall use fire and other disturbance regimes to maintain and enhance forest ecosystem health where appropriate to the forest type or scale (AZ 4.5.3).
- 6.3.11 The contribution of the disturbance regime to the maintenance and protection of biological diversity values shall be reviewed regularly. The results of the review shall be used to modify the disturbance regime in the future in order to increase its effectiveness. (AZ 4.3.7)
- 6.3.12 FME shall plan for and implement effective measures to reduce the extent and impact of unplanned wildfire (AZ 4.4.6).
- 6.3.13 FME shall identify, assess and prioritize any potential damage agents (such as weeds, insect and vertebrate pests, and diseases and pathogens) that may impact ecosystem health and vitality (AZ 4.5.1).
- 6.3.14 Weed, pest, disease and pathogen control plans are implemented to ensure ecological functions are maintained including ecosystem regeneration and succession and species diversity.

6.4 What can be expected from forest certification?

Each forest certification system applies principles, criteria, and indicators to assess the extent of application of sound environmental management practices. Unfortunately, there are few rigorous, on-the-ground studies that evaluate the extent to which forest management consistent with certification standards actually represents sustainable forest management. The evidence therefore depends on the stringency of the standards under the different schemes, and the rigor of the certification process (EEM 2007).

A comparison of the principles and criteria of the FSC, CSA, and SFI certification systems (EEM 2007) suggested that application of FSC standards to forestry comes the closest to sustainable forest management, at least in the case of Canadian forests. This conclusion was based in part on the fact that FSC has rigorous forest management criteria that include the protection of ecologically important forests and the banning of conversion of natural forests into plantations. The EEM study also concluded that CSA certification can be acceptable but further knowledge of the forest management practices would be required to ensure that the required environmental performances are adequately defined and managed. They found the SFI Program to be weaker with respect to forest management practices. Although FSC does not claim that the forests it certifies are 'sustainably managed', the study concluded that it is the scheme most likely to approach sustainable forest management through rigorous performance-based standards (EEM 2007).

SmartWood, one of the certifying bodies for FSC, conducted a study in which they examined the changes that forest operators were required to make during their certification assessments (Newsom and Hewitt 2005). They studied 129 operations in 21 countries (predominantly temperate and boreal but also including tropical operations) and found that the most prevalent environmental requirements (pre-conditions and conditions) for bringing forest management to the level that it could be certified by SmartWood were protection of riparian buffers and improved management of aquatic resources, woody debris, snags and legacy trees, improved treatment of sensitive sites and HCVFs, and improved treatment of threatened and endangered species. The most common required changes in forest management practices were to improve their roads and skid trails, improve their methods for securing natural regeneration and reforestation activities, and use more environmentally sound methods for using chemicals (e.g., herbicides and pesticides). Gullison (2003) also carried out a less intensive study on the FSC certification process using the same database and found similar results, as did Hirschberger (WWF European Forest Programme 2005) for operations in six European countries. Such analysis of Corrective Action Requests (CARs) hinges on the assumption that required changes in management practices will lead to positive effects on biodiversity (see also Jansen and van Benthem 2009; Karmann and Smith 2009) – the assumption that is the subject of this report.

When individual issues related to compliance with certification guidelines were examined, Newsom and Hewitt (2005) showed that the specific actions required of forest operators were diverse and tailored to the particular operation under assessment. In the case of management of riparian zones and aquatic resources, for example, managers were sometimes required to improve their protection of buffer zones by better complying with government regulations and best management practices. In other cases they were required to consult more with local stakeholders and the scientific community about appropriate buffer zone management, and, in a few cases, were given explicit guidance by SmartWood assessors on how their buffer zone management practices should change.

Even for the most stringent of standards applied with the most rigorous audits it must be recognized that certified forests are not pristine, and impacts of certified forestry on biodiversity are inevitable. Certification can be fairly expected to promote forest management that is better for biodiversity (among many other things) than conventional, 'business as usual' logging, but certified forests cannot be expected to be equal to undisturbed forest or to replace all of the biodiversity benefits of full protection (section 4.2). Despite this fairly obvious condition, there is a large body of literature that compares good management practices against undisturbed reference states, as will be seen in the next chapter.

Although all forest certification systems include management prescriptions that are of clear value to biodiversity, Gullison (2003) argued that it does not necessarily follow that certification contributes to biodiversity conservation. Although we found the argument weak, the lack of relevant data is a severe impediment to unbiased analysis. To further the cause of critical evaluation of the biodiversity benefits of certification, in the next section we focus on scientific findings that indicate whether or not certification and associated management activities have a positive influences. We focus on guidelines related to RIL, riparian buffer zones, green tree retention, protected areas, HCVFs, and corridors because these were most consistently associated with certification (Gullison 2003; Newsom and Hewitt 2005; Rametsteiner and Simula 2003).

In a similar vein, Rametsteiner and Simula (2003) declared that "after 10 years of implementation, it is evident that the original intention to save tropical biodiversity through certification has largely failed to date. Most of certified areas are in the temperate and boreal zone, with Europe as the most important region. Only around ten per cent is located in tropical countries".





7

Findings of the review

In this chapter we review the available (scientific) literature on the effects of forest certification and good forest management practices on biodiversity. First we present studies making a direct comparison of biodiversity before and after certification of forest management. Only few studies were found. Subsequently we present the effects on biodiversity of seven good forest management practices which are frequently applied in certified forests: reduced-impact logging, riparian buffer zones, green tree retention, protected areas, management associated with High Conservation Value Forests, and corridors.

We present literature for tropical, boreal and temperate zones separately because of the different forest management systems in these areas. Except for tropical studies, it was not always clear in which forest type a study was done. Therefore we distinguished boreal and temperate studies based on the countries in which they were performed, with Scandinavia, Estonia and Canada as boreal zones, and the USA, Australia, Europe and Japan as temperate zones.

In many cases the articles reviewed contained little information about the management system which was applied, except for clear cut forests, but we feel safe in assuming that all tropical studies involved selective logging of unlogged or lightly logged natural forests. Where plantations were studied (just three studies), this is clearly indicated.

The literature review on direct certification effects is complete as far as we are aware. At least for RIL in tropical areas and for riparian buffer zones, green tree retention and corridors in boreal/temperate areas, the literature overview is not exhaustive. Rather we present the key studies on these topics which adequately summarize the present state of knowledge.

7.1 Direct effects of certification

A forest manager who wishes to receive a certificate of 'good' forest management has to comply with the standards of the relevant certification system. With respect to biodiversity conservation, these standards require conserving as many of the features of the undisturbed forest as possible in areas from which trees are harvested.

The expectation of complying with these certification standards is that forest biodiversity in these forest management units will be higher than in similar forests which are not certified.

7.1.1 Tropical zone

A lowland dipterocarp forest in Malaysian Borneo (Sabah) that was certified by FSC in 1997 was studied by Lagan *et al.* (2007). In this case, the most important measures implemented for 'good forest management' in the study area were RIL and the protection of HCvFs. The authors concluded that certification had a positive impact on biodiversity because the certified forest sustained denser populations of endangered large animals including orangutans and elephants than elsewhere in Sabah, but no actual data were presented in their paper. Similarly, studies conducted in the same area on plants and soil macrofauna showed that use of RIL had biodiversity benefits. For instance, tree species diversity was equally rich in the old-growth forest and in the forest harvested using RIL techniques, where climax forest and important commercial timber species of Dipterocarpaceae dominated, but was much lower in the forest harvested using conventional methods. Moreover, the size structure of canopy-tree populations showed that dipterocarp trees regenerated well in the forest harvested by RIL. In contrast, pioneer species dominated the forest harvested by the conventional method.

7.1.2 Boreal zone

Probably the most extensive pre/post certification research in which a number of field-level indicators were compared in the boreal region was done by Sverdrup-Thygeson *et al.* (2008) in Norway. Over 200 pre- and post-certification (PEFC) regeneration areas (mostly clear-cuts) were analyzed for implemented biodiversity measures. In the post-certification areas more green retention trees (trees left during harvesting to provide live and dead/decaying wood as habitat) were present and the buffer zones along rivers, bogs, and lakes were wider than in pre-certification areas, which are expected to benefit biodiversity (but these benefits were not measured directly). However, 21% of the post-certification areas did not have sufficient retention trees to comply with the certification standard. Concerning the management of small swamp forests and the damage done due to off-road transport, little improvement was seen.

Mielikäinen and Hynynen (2003) surveyed the effects of forest certification (PEFC) by analyzing existing literature about Finland (in Finnish). They found that by implementing management practices such as planting of a variety of species and thereby creating mixed stands, protecting small-sized valuable habitats in commercially managed stands, and leaving retention trees all are expected to have positive impacts on the biodiversity in boreal forests (not supported with data).

7.1.3 Temperate zone

No studies found.

7.1.4 Summary

Few studies on pre- and post certification forest biodiversity have been carried out to date. The results have been summarized in table 9. The one study we found was performed in the tropics (Lagan *et al.* 2007), focused on the combined direct effects of RIL and HCvF, but did not support their findings with data. One of the two reviewed studies conducted in boreal forests (Sverdrup-Thygeson *et al.* 2008) did

not directly measure the effects of certification on biodiversity but instead assessed whether certain measures prescribed for sustainable forest management were being executed. The other article (Mielikäinen and Hynynen 2003) also did not include data on biodiversity benefits but instead relied on a review of the existing Finnish literature on certification. No studies were found from the temperate zones.

Table 9. Summary of the direct effects of certification on different taxa.

Certification	Comparison of certified forest landscape with reference managed forest landscape	Comparison of certified forest landscape with undisturbed forest landscape	Tropical	Temperate	Boreal
Forest biodiversity	+?	n.d.			x
Mammals	+		x		
Trees	+	=	x		
Soil macrofauna	+?				
Landscape features	+				x

+ (=, -) means that application of certification or a good management practice resulted in a positive (insignificant; negative) effect on biodiversity compared with the reference. N.d. not determined.

7.2 Reduced-impact logging (RIL)

The FAO defines reduced-impact logging (RIL) as: ‘Intensively planned and carefully controlled implementation of harvesting operations to minimize the impact on forest stands and soils, usually in individual tree selection cutting’. It generally includes, but is not limited to, pre-harvesting inventory and mapping of individual crop trees, pre-harvesting planning of roads, trails and landings while minimizing disturbance and their construction adheres to engineering and environmental guidelines, pre-harvest liana cutting where lianas inter-connect tree crowns, felling and bucking techniques to minimize damage, careful yarding and conducting a post-harvest assessment to provide feedback to the concession holder and logging crews (FAO 2004; Meijaard *et al.* 2005). When RIL techniques are not implemented, the volume of commercial timber in the second and third cuts is expected to be much reduced compared to the first cut (FAO 2004).

RIL is supposed to preserve more of the characteristics of the undisturbed forest than conventional logging. On average, logging gaps are expected to be smaller, the damage to the remaining stand and the soil will be reduced, and the microclimate will be less affected than in the case of conventional logging. If more of the features of the undisturbed forest are conserved, then this should directly and indirectly conserve species and individuals and reduce the influx of pioneer and non-forest species. It is therefore expected that in forests harvested using RIL, more species and individuals characteristic of undisturbed forest communities will be present compared to conventionally logged forest, but not more than in undisturbed forests. Because of the change in habitat conditions due to opening of the canopy, not only species abundance but also community composition might change when compared to the undisturbed forest.

Using the literature survey, we evaluated the assumption that forest management units harvested using RIL retain higher levels of biodiversity than comparable forests harvested conventionally.

7.2.1 Tropical zone

Large numbers of studies are available on RIL in the tropics.

Timber harvests, whether or not RIL is applied, affects bat diversity by altering the composition and structure of vegetation, availability of food resources, numbers or types of refuges and roosts, and microclimate and disturbance regimes of ecosystems. Bats could be keystone species as they are primary agents of pollination and seed dispersal for many pioneer plants. Castro-Arellano *et al.* (2007) investigated the effect of low harvest RIL on bats in a natural forest in the Brazilian Amazon. The effects were assessed 2-4 years post-harvest. It appeared that only 5 of 15 species showed significant responses to RIL, of which only two species (frugivores) decreased in numbers. Some rare species that were present in the control forest were absent in the cut forest but evidence is lacking to conclude this is resulted from logging. Overall bat species showed less response to RIL than to CL or forest conversion. The authors concluded that even though RIL had minor effects on biodiversity in the short-term, long-term effects might be different if the pollinating and seed dispersing services of bats are disrupted.

Presley *et al.* (2008) also studied the effects of RIL on bat populations in the Brazilian Amazon, 2 to 4 years after logging. They found that the most abundant species were not affected by the disturbance. In contrast, more uncommon or rare species were absent in logged forest as compared to the undisturbed forest. No comparison to a conventionally logged forest was made. They suggest that arrangement of RIL sites in a matrix of undisturbed forest may allow source-sink dynamics to mitigate the deleterious effects of RIL on rare or sensitive species and thereby enhance sustainability at a regional scale.

Felton *et al.* (2008a) investigated the response of birds to RIL in a certified (FSC) forestry concession in Bolivia that was logged 1-4 years previously. They found the composition of the bird community to differ largely between logged and unlogged parts of the forest. In unlogged parts over 40 % of the species that they found were sensitive to disturbance, have high conservation value. Most of these species were absent in logged areas, or were present in very low abundance. The majority of the birds found with higher abundance in the unlogged areas of the concession, were associated with forest habitats dominated by large trees, or a high diversity of trees, providing dense canopy cover and deep leaf litter, with an understory dominated by ferns. These conditions are not maintained in logged areas. Given that in the forest they studied, skid trails, roads and landings disturbed 25 % of ground area with an additional 25 % of the canopy opened due to tree felling (Jackson *et al.* 2002), even purportedly RIL operations can result in substantial disturbances. Furthermore, logging gaps were larger than natural tree-fall gaps. The introduction of logging roads and skid trails, as well as the increased frequency and extent of tree-fall gaps, all contribute to a reduction in canopy continuity and the increased prevalence of forest area in early successional stages. Felton *et al.* (2008a) concluded that although

RIL may cause less damage to the forest than conventional logging, in the forest concession in which they worked, the goals of sustainable forest management were not being reached because some bird species were adversely affected⁷. This study, like several others, was apparently designed with the intention to show a deleterious impact of logging. This bias derives from the researchers' focus on only the most severely disturbed areas such as logging roads and skid trails. Furthermore, the authors disregarded several studies conducted by local researchers in the same area that came to different conclusions.

In a publication that overlaps substantially with the one discussed above, Felton *et al.* (2008b) compared bird communities in logging-induced and natural tree fall gaps in the same FSC certified forest in Bolivia 1-4 years after logging. Natural tree-fall gaps supported higher bird species richness than the anthropogenic tree-fall gaps. Bird community composition was also significantly different between natural tree-fall gaps and logging gaps in this concession. Furthermore, 50% of the bird species associated with natural tree-fall gaps are considered to be relatively vulnerable to human disturbance. This indicates that natural-tree fall gaps support bird species of higher conservation importance than anthropogenic gaps. The gaps created by RIL were quantifiably different from gaps caused by natural processes. Logging gaps were larger, differed in microclimate, were significantly lower in understory density, and differed in the composition of regenerating vegetation. Differences in bird community composition and abundance within logging gaps may therefore be the direct result of either microclimatic differences, and/or the associated changes in vegetation structure. These differences were perhaps further accentuated by pre-harvest liana cutting. As lianas provide a distinct foraging substrate, obligate or facultative liana foragers may be detrimentally affected by this practice. As the practice of pre-harvest liana cutting is successfully reducing liana loads in logging gaps, it is possible that this practice influences the availability of suitable habitats within natural and anthropogenic gaps. The differences between gaps are directly relevant to those considering the compatibility of logging with biodiversity maintenance.

Wunderle *et al.* (2006) studied the effect of RIL on birds in the Brazilian Amazon, 20-42 months after logging. They found that species richness did not differ between control and cut forests blocks. Total captures however, were higher in cut than in control forests as well as captures of nectarivores, frugivores and insectivores. Increased captures of nectarivores and frugivores following timber harvest were expected, given the reliance of these guilds on food resources that are widely dispersed and often patchy, or available for a short period in early successional

7 The interpretations of RIL impacts on bird communities by Felton *et al.* (2008a&b) highlight a few fundamental problems. First of all, they compared logged with unlogged forests, not RIL with conventionally logged areas where the direct as well as the indirect effects of logging (e.g., hunting) might have been much more severe. More subtly, their repeated reference to this "RIL forest" belies the fact that there were serious breaches of RIL guidelines by logging crews working in this forest (Jackson *et al.* (2002) FEP, pers. obs., RZ pers. obs.). At the time, the certification auditors were aware of these problems, set stringent conditions for maintenance of certification, and logging damage was subsequently reduced. More fundamentally, RIL crews will always vary in their skill levels, the incentives they receive for treating the forest gently (or penalties from not doing so), and the resulting impacts they have on harvested stands.

patches in logged-over forest. Consequently, some nectarivores and frugivores may be pre-adapted to take advantage of post-harvest increases in flowering and fruiting that are associated with increased light levels after tree-felling. Not surprisingly, members of these guilds frequently are resistant to logging, at least in the short-term (for references see Wunderle *et al.* 2006). Higher capture rates of insectivores in cut vs. control forest were inconsistent with findings that understory and terrestrial insectivores are especially sensitive to timber harvest. This sensitivity has been attributed to the specialized foraging modes of insectivores, which often require open forest understory, as well as postharvest changes in prey availability. The authors suggest that insectivore declines might not occur until several years after tree-felling. In general, they found that the effects of logging were relatively minor and that low harvest rates and reduced-impact methods may help to retain aspects of avian biodiversity in Amazon forest understories.

Davis (2000) studied whether RIL helps maintain dung beetle biodiversity in a tropical rainforest in Borneo. Sites were logged in 1993: one conventionally and one with RIL. Results were compared with primary forest and an old conventionally logged forest (1981). Of the two 1993 sites, the higher diversity and species richness was recorded in the RIL forest. Besides this, the RIL forest contained some primary forest species that were lacking in the conventionally logged forest. The dung beetle community in the 1993 conventionally logged forest was similar to that in the 1981 conventionally logged forest, indicating little recovery over this time period. This study suggests that RIL served to better preserve the primary forest beetle assemblage than conventional logging.

Azevedo-Ramos *et al.* (2006) studied the short-term (6 months after logging) effects of reduced-impact logging on Amazonian (Brazilian) ants, arachnids, birds, and mammals. Overall, the faunal effects of RIL were minor. The major changes were an increase in species richness of ants, arachnids, and birds, confirming the expected pattern after recent disturbance. Mammals showed no changes in richness, abundance, or composition. In contrast, invertebrates responded quickly to logging, showing changes in abundance (arachnids), composition (arachnids), and richness (arachnids and ants). Several new groups of ants and arachnids were recorded after logging, indicating that new habitat conditions (e.g., gaps, light, heat) caused by disturbance may have created new opportunities for colonization or establishment of previously absent or rare species. The low intensity of logging and the connectivity with a matrix of logged and unlogged forests, may have promoted rapid recolonization of the study sites. The lower species loss in RIL forests compared to other types of land use in Amazonia highlights the value of this technique for conservation purposes among prominent economical activities.

Studies done in a Malaysian FSC-certified forest showed that RIL practices had a positive influence on the richness of species and families of canopy trees as compared to the conventionally logged forests. Species richness and composition was not different from that in the nearby pristine forest. For soil macro-fauna, the density and the richness of taxonomic groups at the order or equivalent taxonomic level (not species in this case) did not differ among the RIL, conventionally-logged,

and pristine forests. In contrast, composition of the soil macro-fauna community was modified greatly by conventional logging, but less so by the RIL operation. Therefore, RIL could maintain the richness, density, and composition of soil macro-fauna reasonably well at least at higher taxonomic levels. RIL did not maintain populations of flying insects (fruit flies, bees, sap beetles and others) at a level equivalent to the pristine forest, but maintained a higher abundance than conventional logging. A limited survey using camera traps indicated that the number of mammal species was greater in the RIL forest than in the conventionally logged forest. A few mammal species demonstrated a higher frequency of appearance in the RIL forest than in pristine forest. Large mammals are often hunted for bush meat unless the access of hunters is physically limited. The greater species variety and population abundance in the RIL forest may just reflect the protection from hunting, because the access to the RIL forest is limited by locked gates. However, another independent census on the orangutan population from a helicopter also indicated a significantly higher nest density in the RIL forest than in the surrounding forests (Mannan *et al.* 2008).

Tropical forest silviculturalists have long been aware of the negative impacts of lianas on timber production. More recently, ecologists have become aware of the importance of lianas to forest biodiversity and ecosystem functioning (e.g., Mason and Putz 1991). Research on the effects of RIL (with pre-cutting of climbers) on liana species in the Brazilian Amazon 10 years after logging revealed that lianas respond differently to different logging practices (conventional logging and RIL) (Gerwing 2006). The response depends on their reproductive mode. Conventional logging is likely to favor species that sprout profusely from fallen and prostrate stems while pre logging climber cutting favors species that sprout from stumps. Even though RIL with liana cutting causes less damage than conventional logging, liana species composition changes after disturbance as compared to the undisturbed forest (Gerwing 2006). Balancing the needs of timber production and forest conservation will require a more detailed understanding of the responses of individual liana species to silvicultural treatments as well as of the roles of different liana species in biodiversity maintenance.

Kukkonen *et al.* (2008) examined the suitability of tree fall gaps as regeneration sites for commercial tree species in certified (FSC), conventionally managed and natural forests in Honduras. What they found was that, in support of their hypothesis, gaps in certified forests showed lower levels of logging disturbance than gaps in conventionally logged forests. Contrary to their expectations, higher regeneration of timber species was found in conventionally logged forests than in certified forests. The authors suggest that logging in the certified studied gaps might have been more intensive in the past, leading to a scarcity of timber seed trees. This finding might also relate to the suggestion made by Fredericksen and Putz (2003) that for some forests and some species, intensifying logging practices might better serve to provide sustainable timber yield, since some timber species require more disturbed gaps for seed survival and growth.

Sist and Ferreira (2007) studied the sustainability of RIL in a FSC-certified forest in the Brazilian Amazon. The sustainability of timber management applying RIL

was evaluated through the calculation of the recovery level of commercial trees in different scenarios. In the most optimistic scenario (growth rate of 5 mm year⁻¹ and 1 % annual mortality), after 30 years, only 50 % of the commercial stand would recover, provoking a drastic reduction of the harvesting intensity at the second felling cycle. Within a 30-year felling cycle (i.e. the legal felling cycle duration in the Brazilian Amazon) and even under RIL systems, the present logging intensity occurring in the study area (6 trees ha⁻¹) is not compatible with sustainable yield production on a long-term basis. This study showed that in the Amazon, RIL alone is not sufficient to achieve sustainable forest management. More elaborate silvicultural systems must be urgently elaborated and implemented to ensure that the forest will still be sustainably managed on a long-term basis.

Logging influences natural processes such as gene flow, mating system, genetic drift, and spatial genetic structure, both directly and indirectly. Logging reduces tree species density in the reproductive population which causes an increase in the distance between reproductive trees. It may also create subpopulations in which the capacity for exchanging genetic material decreases. Moreover, logging causes impacts on the early ontogenic stages of a population as a substantial number of seedlings, juveniles and unlogged trees can be killed or damaged during harvesting operations or afterwards. The consequences can be negative for the reproductive success and gene flow of not only tropical trees but also other organisms.

Lacerda *et al.* (2008) studied the effect of RIL on genetic diversity of a logged tree population in the Brazilian Amazon. Their results show that RIL causes the reduction of genetic diversity and mainly the loss of alleles. The harvest of 61 % of the reproductive trees caused a loss of 25 % low frequency and rare alleles from the mature population. However, part of the "lost" alleles are present in juvenile and sub adult populations which might allow their reintroduction into the population when these trees become reproductive. The loss of rare alleles can have an impact on the long-term genetic fitness of a population, which might be of particular importance with the current tendencies of higher global temperatures related to climate change. In this sense, some rare alleles can represent the genetic potential required for a population to adapt through natural selection; current high-frequency alleles might have been the result of previous selection in past and current environments. However, despite the potential allelic recovery, the results indicate that selective logging affected the gene pool of the population.

Pinard *et al.* (2000) found that in Malaysia RIL significantly contributed to a reduction in stand damage and damage to the soil. Residual stands in RIL areas had greater vertical structure than conventionally logged areas with positive gains for conservation of biodiversity (but this was not measured directly).

Rockwell *et al.* (2007) concluded that RIL in Brazil (in a FSC certified forest) decreased damage to residual trees compared to conventional logging (also found by Feldpausch *et al.* 2005), but damage due to skid trails remained, and the forest gap mosaic was substantially altered compared to undisturbed forests.

The FAO compiled studies on RIL and concluded that RIL caused less damage to residual trees than conventional logging and generated considerably less wood waste (FAO 2004 and references therein).

7.2.2 Boreal zone

No studies found.

7.2.3 Temperate zone

No studies found.

7.2.4 Summary

All the studies we reviewed on the effects of RIL were carried out in the tropics (Table 10). Forests harvested with RIL reportedly have more species and individuals than forest harvested with conventional logging techniques in most of the studies we reviewed (Castro-Arellano *et al.* 2007; Davis 2000; Felton *et al.* 2008a; Gerwing 2006; Mannan *et al.* 2008; Wunderle *et al.* 2006). The forest is damaged to a lesser extent than in the case of conventional logging (Pinard *et al.* 2000; Rockwell *et al.* 2007). As a result, more of the original habitat is maintained, which benefits the structural complexity of the forest and many species dependent on the maintenance of pre-intervention conditions.

The responses of individual (groups of) species to RIL are highly variable. This can be attributed to a number of factors. Short-term responses can be very different from long-term responses, due to the temporary availability of ephemeral habitats and associated resources, or time-lags in the emergence of effects in, e.g., long-lived species. Different functional groups may be responding in a different way, as resources and habitat features required by them respond in different ways to RIL. The response may depend on which biodiversity attribute is considered: responses in terms of abundance may be different from responses in composition or diversity. Finally, the process considered influences conclusions. Impacts on reproduction may be different than on pollination, regeneration, growth or dispersal of species. This variability makes it difficult to identify general trends.

When compared to undisturbed forests, often the most abundant species are still present in RIL forests, albeit sometimes in lower numbers. The reduced abundance may, as some authors speculate, be a temporary phenomenon (Wunderle *et al.* 2006). Some researchers even observed increases in species richness and numbers of individuals, for instance for arachnids, ants, some birds and some mammal species (Azevedo-Ramos *et al.* 2006; Mannan *et al.* 2008; Wunderle *et al.* 2006). The increase mainly has to do with (often temporary) changes in habitats or resource availability, for instance nectarivorous and frugivorous birds benefit from increased flowering and fruiting of trees after harvest.

When compared to unlogged forests, RIL has some negative impacts on some species of plants and animals. In some cases rare or threatened species, or species with otherwise high conservation value that are present in undisturbed forests, are present in lower numbers or even absent in logged forests, as was the case for bats,

birds and beetles (Castro-Arellano *et al.* 2007; Davis 2000; Felton *et al.* 2008a; 2008b; Presley *et al.* 2008). Considering the high conservation value of such species, their decrease in abundance seriously affects biodiversity.

The species composition in the forest was often reported to change due to RIL compared to undisturbed forest (Castro-Arellano *et al.* 2007; Felton *et al.* 2008a; 2008b; Gerwing 2006; Mannan *et al.* 2008). This does not necessarily change species richness, but it may change biodiversity when species that were originally present (some of which may be rare) are replaced by different species typical of disturbed habitats.

The studies we reviewed were mostly focused on the first 4 years after timber harvesting. Some studies have indicated that more research is needed to determine the long-term effects of RIL on species (Castro-Arellano *et al.* 2007; Gerwing 2006). Some studies indicated that the current RIL practices do not guarantee full retention of species prior to logging (Felton *et al.* 2008a) or sustained timber yields (e.g., Dauber *et al.* 2005, Sist and Ferreira 2007, Putz and Zuidema 2008) and that they need immediate improvement.

Table 10. Summary of the effects of reduced - impact logging on different taxa.

Reduced Impact Logging	Comparison of managed forest landscape (RIL) with reference managed forest landscape	Comparison of managed forest landscape (RIL) with undisturbed forest landscape	Tropical	Temperate	Boreal
Biodiversity	n.d. but a positive effect is plausible	n.d. a negative effect is plausible	x		
Mammals	+ (bats, Castro-Arellano et al 2007) + (mammals, Mannan 2008)	=/- (bats, Castro-Arellano et al 2007 ; Presley et al. (2008) = (mammals, Azevedo-Ramos et al. (2006) + (mammals, Mannan 2008)	x		
Birds		- (Felton 2008a,b) = (richness, Wunderle 2006) + (abundance, Wunderle 2006 + short term effects on birds, Azevedo-Ramos et al. (2006)	x		
Invertebrates	+ (dung beetles, Davis 2000) + (soil macrofauna, Mannan 2008) + (flying insects, Mannan 2008)	=/- (dung beetles, Davis 2000) + (short term effects on arachnids and ants, Azevedo-Ramos et al. (2006) = (soil macrofauna, Mannan 2008) - (flying insects, Mannan 2008)	x		
Plants	+ (trees, Mannan 2008) - (tree regeneration Kukkonen et al. (2008)	= (trees, Mannan 2008) - (lianas, Gerwing 2006) - (commercial trees, Sist and Ferreira (2007) - (tree genetic diversity, Lacerda et al. (2008)	x		
Landscape features	+ (many authors)		x		

+ (=, -) means that application of certification or a good management practice resulted in a positive (insignificant; negative) effect on biodiversity compared with the reference. N.d. not determined.

7.3 Riparian buffer zones

Riparian buffer zones, sometimes called stream corridors, refer to the land-water interface that extends from the stream through the riparian zone to the adjacent upland. These zones hold unusual promise for biological conservation because diversity of plant and animal species in riparian areas is often disproportionately high. Riparian buffer zones also connect and interface with other ecosystems which may facilitate high levels of ecological and genetic exchange (see also section 7.7). Mammals, birds and plants probably use these zones as habitat connectors, travel corridors and also as refugia (Spackman and Hughes 1995).

Protecting riparian buffers is thought to reduce forestry-related biodiversity loss in forest management units in a number of ways: by protecting specific riparian habitats and their biodiversity, by reducing logging impacts (sedimentation, temperature changes etc.) on streams and rivers, and by providing corridors between habitats that serve as refugia for plants and animals that are sensitive to logging. Therefore, it is expected that forest management units containing riparian buffers will have higher levels of forest-based biodiversity than similar forest management units lacking them. In the absence of direct measurements of biodiversity, a positive effect is likely once it can be shown that undisturbed streamside and aquatic habitats have been maintained in logged forests, and if evidence for the corridor function of riparian buffers can be demonstrated.

7.3.1 Tropical zone

To date, almost no studies in the tropics have focused on the importance of riparian zones along forested streams in logged forests. Given current rates of forest loss, a better understanding of the importance of these habitats might help inform conservation management (Chan *et al.* 2008).

Chan *et al.* (2008) compared the abundance and assemblage composition of birds and the biomass of insects between the riparian zone of a small stream and sites further inland within a secondary forest in Hong Kong, situated at the monsoonal northern margins of the Asian tropics. The results of the study suggests that because of the higher insect biomass found near streams as compared to further away, riparian zones associated with small tropical streams are important (feeding) habitats for forest birds. Consequently, degradation of riparian forest may lead to large disturbances to terrestrial bird assemblages, notwithstanding the small areas that riparian zones occupy. Riparian buffer strips, which are beneficial for birds in temperate regions are currently underused as a management tool in the tropics but could be useful for the conservation of riparian birds and other wildlife. They should be a priority for conservation planning and, justifiably, are emphasized in most certification criteria and indicators.

7.3.2 Boreal zone

Whitaker *et al.* (2000) compared the abundances of flying insects along undisturbed lakeshores and riparian buffer strips in disturbed areas in balsam fir (*Abies balsamea*) forests in Canada. Capture rates of insects were consistently higher along riparian

buffer strips than along undisturbed shorelines. The authors hypothesize that riparian buffer strips provide shelter from strong winds, and thereby act as collecting sites for insects blown in from exposed clear-cuts and lakes. As a result they represent high-quality feeding habitat for aerial foraging and foliage-gleaning insectivorous bird species that are not restricted to specific habitat types. Elevated densities of insect prey, combined with high habitat diversity, may lead to the high density and diversity of breeding-bird assemblages along riparian buffer strips (results from a concurrent parallel study by Whitaker). Other insectivorous wildlife, such as bats, spiders, and dragonflies, which forage heavily on localized concentrations of flying insects along forest edges, may also treat buffer strips as high-quality habitat patches (for references see Whitaker *et al.* 2000). However, the authors state that caution must be exercised in extrapolating local increases in abundance and habitat quality to the landscape or population level. The net effect of habitat loss through clear-cutting and fragmentation, as well as possible negative effects on reproductive success, may exceed any local benefits a species derives from buffer strips.

Mönkkönen and Mutanen (2003) studied the utility of riparian corridors in boreal forest landscapes as habitats and dispersal routes for forest-associated moths in Finland. They investigated their occurrence and abundance in four habitat types—forest interior, forest edge, forested corridors, and clear-cuts—along riparian zones of 30-70 m wide. The number of species and total number of individuals did not differ among the forested habitats (interior, edge and corridor) but were significantly lower in clear-cuts. These results suggest that corridors in boreal forest landscapes serve as breeding habitats or dispersal routes for the moths and may direct the movements of these forest-associated species. The authors emphasize that results from studies showing positive effects of corridors on dispersal or population persistence should not, however, be used to justify more habitat destruction.

Given that land snails are generally poor dispersers, the short-term effects of a disturbance are important for their survival in the long term. In other words, local extinctions may lead to extended periods of absence. Hylander *et al.* (2002) investigated the short-term effects (2.5 years after logging) of forest clear-cutting on land snails (terrestrial gastropods) along small streams in Sweden. They compared sites that were clear-cut and sites that had a riparian buffer zone with undisturbed forests. Most species were negatively affected by clear-cutting. Most of the results from the buffer strips were intermediate between the reference site and the clear-cut sites in terms of the change in number of snail species and in overall abundance. This suggests that the 10-m buffers functioned, but that they were too narrow to retain an unaffected snail community.

Barton *et al.* (1985) studied the relationships between riparian land use and environmental parameters that define the suitability of southern Ontario (Canada) streams for trout. The only environmental variable which clearly distinguished between trout and non-trout streams was water temperature: streams with a temperature less than 22° C had trout; warmer streams had, at best, only marginal trout populations. The most important cause of high stream temperatures is direct insolation resulting from the absence of shading by a forest canopy. Control of

temperature can be achieved through establishment or maintenance of forested riparian buffer strips, where both width and length seem important.

7.3.3 Temperate zone

Spackman and Hughes (1995) censused bird, mammal and vascular plant species in 200-m long plots at varying distances from six mid-order streams in the USA to determine how wide corridors need to be to conserve biological richness. They found that no standard minimum corridor width could be identified. For instance, to include 90% of the stream side plant species, minimum corridor width ranged from 10 to 30 m depending on the stream, and width varied from 75-175 m to include 90% of the bird species. Edge effects and the surrounding landscape are likely to impact streamside communities and appropriate dimensions of stream corridors. The authors conclude that use of a standard corridor width to conserve species is a very poor substitute for individual stream specific assessments of species distributions.

Hanowski *et al.* (2003) tested the effect of clear-cut and partial cutting in adjacent forest on bird communities in the riparian zone in the USA. Bird surveys were completed 1 year prior to, and 3 years after harvest and revealed that when no harvesting occurred in the buffer zone, the bird community in the zone did not seem to be affected by harvesting practices. When harvesting did occur within the riparian zone, breeding bird composition included more early-successional species than plots in which no harvesting occurred.

Lloyd *et al.* (2006) studied bat activity in riparian buffer zones in an Australian timber production forest. They found no differences in the riparian zone of logged forest and mature forest. Bat activity, foraging rates and species richness were similar to each other in all buffered streams. They found that species richness is not different; however, they do not present data on species composition in the different forest types. Based on their findings the authors conclude that creating riparian buffer zones is a suitable management practice to protect the biodiversity of –at least- the riparian areas.

Crawford and Semlitsch (2007) in the USA estimated the minimum width of the riparian buffer zone needed for the protection of stream-breeding salamanders. It is currently set to 9-30 m depending on the size of the stream. A width of 27 m encompassed 95 % of the salamander assemblage and when taking edge effects into account an additional 50 m would be advisable, yielding a total buffer of 77 m. When all species are considered, the maximum buffer zone needed was 93 m. To protect stream amphibians and other wildlife dependent on riparian areas, land managers and policy makers must consider conserving more than aquatic resources alone. Developing core terrestrial habitat estimates and buffer zone widths for wildlife populations is a critical first step in the conservation of many semi aquatic organisms and protecting biodiversity.

Johnston and Frid (2002) studied the effect of riparian buffer zones on salamanders in the USA and found that salamander movement in riparian strips was not different

from that at forested sites, but was different from that at clear-cut sites. According to the authors it is important to note that some microclimatic gradients extend up to large distances into the forest from the clear-cut edge and that in such cases wide buffer-strips may be necessary to maintain natural riparian microclimatic gradients intact.

Goates *et al.* (2007) tested whether a buffer zone of 35 m adjacent to streams and wetlands in the USA was sufficient to protect the habitat of a boreal toad species. A buffer zone of 30.5m is commonly used by resource managers to protect species in riparian and wetland systems. This standard was developed to protect water quality, not biodiversity. It was found that the standard implementation of 30.5 m buffers did not protect all critical habitats for the toads: many small streams and seeps used by toads were outside of buffer zones. Managers should consider several factors when establishing buffer zones: buffer requirements may vary by time of year, season and sex. This suggests that each site should be evaluated for its needs individually, in different seasons over several years and that this information should be used to tailor buffers zones for that site.

Burke and Gibbons (1995) investigated whether the delineation of wetlands (established by the federal state) and the buffer zone around wetlands (established by some state and local jurisdictions) in the USA are sufficient to protect freshwater turtles. Two critical life-cycle stages, nesting and terrestrial hibernation, occurred exclusively beyond wetland boundaries delineated under federal guidelines. The most stringent state buffer zone insulated 44% of nest and hibernation sites, the remaining sites were not protected (but not necessarily damaged either). This study indicates that freshwater turtles required a 275 m buffer zone to protect 100% of the nest and hibernation sites. Insulating 90% of these sites required a 73 m buffer zone. Thus in order for buffer zones to be useful, they need to be sufficiently large.

Roth (2005) assessed the importance of buffer zones to a snake species in the USA. He monitored males, gravid females and non-gravid females and found that 83% of the observations were done within 10 m of the stream. Population subunits exhibited different patterns of spatial use. Gravid females provided most of the distant observations, inhabiting the surrounding terrestrial habitat up to 94 m from the shoreline. These results show the need for a buffer zone around riparian ecosystems, and highlight the importance of considering spatial use differences between population subunits when outlining buffer zones for conservation purposes.

Semlitsch and Bodie (2003) performed a literature survey and provide an estimate of the biologically relevant size of core habitats surrounding wetlands for amphibians and reptiles. Nearly all studies were done in the USA. They summarize data for 19 frog, 13 salamander, 5 snake and 28 turtle species. Core terrestrial habitat ranged from 159 to 290 m for amphibians and from 127 to 289 m for reptiles from the edge of the aquatic site. Data from these studies indicated the importance of terrestrial habitats for feeding, overwintering, and nesting, and, thus, the biological interdependence between aquatic and terrestrial habitats that is essential for the persistence of

populations and to maintain biodiversity. The minimum and maximum values for core habitats, depending on the level of protection needed, can be used to set biologically meaningful buffers for wetland and riparian habitats.

It is not surprising that the terrestrial ecology of semi-aquatic species is often underappreciated or overlooked by managers and conservation planners. Some semi-aquatic reptiles make only brief visits to terrestrial habitats when nesting, and hibernacula are rarely observed. Additionally, many pond-breeding amphibians are burrowing and rarely observed in terrestrial habitats. Surveys and studies of these animals are consequently concentrated within stream and wetland sites, where they are found seasonally, rather than in terrestrial habitats, where detection is extremely difficult but where much of their life history occurs. Aquatic habitats may not be used by semi aquatic species for extended periods of their lives, including between breeding seasons and during droughts.

7.3.4 Summary

There is substantial evidence that maintaining buffer zones has a positive effect on certain species (Table 11). However, few studies address the direct interest of this report by providing an answer to the question of whether managed landscapes that include riparian buffer zones (such as in certified forest management units) maintain higher levels of biodiversity than managed landscapes without such zones. Instead, many studies demonstrate that riparian zones contain associated certain species, and that preserving buffers likely contributes to their continued existence in disturbed or managed landscapes. This is a related conclusion, but not the same, because it refers to the scale of patches (buffer zones) rather than the full landscape.

All but one study we reviewed were carried out in boreal or temperate regions and many of them focused on amphibians and reptiles. All studies concluded that riparian buffer zones contain important species and some report that in the case of logging, these zones protect species richness and abundance. However, many studies conclude that compared with undisturbed conditions, most current buffer zones are too narrow to preserve most or all of the individuals living in and near streams (Burke and Gibbons 1995; Crawford and Semlitsch 2007; Goates *et al.* 2007; Hylander *et al.* 2002). For instance, in the case of a salamander assemblage in the USA, the width of the zone needs to be 3 to 10 times as wide as it is currently required to protect all species (Crawford and Semlitsch 2007).

The width of streamside buffers needed is stream specific (Spackman and Hughes 1995) and varies with species; required widths range from 127 m to 290 m for amphibians and reptiles as reported in a review study by Semlitsch and Bodie (2003). The required width strongly depends on the behavior of the species in question, which relates to the time of year, season and sex (Goates *et al.* 2007; Roth 2005; Spackman and Hughes 1995). The wider the zone, the more species and individuals are protected (Crawford and Semlitsch 2007; Roth 2005; Spackman and Hughes 1995). Barton *et al.* (1985) report that not only width, but also buffer length is important: the presence or absence of canopy cover upstream affects

environmental conditions further downstream. Because of the variable behavior of species and the variable characteristics of streams, more research is suggested (Semlitsch and Bodie 2003; Spackman and Hughes 1995).

Implementing riparian buffer zones has positive effects on the maintenance of species and individuals, but one should not forget that these benefits do not entirely compensate for the negative effects of habitat loss due to logging (Monkkonen and Mutanen 2003; Whitaker *et al.* 2000).

Table 11. Summary of the effects of riparian buffers on different taxa.

Riparian buffer zones	Comparison of managed forest landscape (with buffer zones) with reference managed forest landscape (no buffer zones)	Comparison buffers in managed forest landscape with undisturbed forest landscape	Comparison of buffer zone with upland habitat	Tropical	Temperate	Boreal
Birds	+? (Whitaker et al. (2000) + (Hanowski et al. (2003)	= (Hanowski et al. (2003)	+ (Chan et al. 2008)	x	x	x
Invertebrates	+? (Whitaker et al. (2000) + (moths Mönkkönen and Mutanen (2003) + (snails, Hylander et al. (2002)	- (Whitaker et al. (2000) = (moths Mönkkönen and Mutanen (2003) - (snails, Hylander et al. (2002)	+ (Chan et al. 2008) = (moths Mönkkönen and Mutanen (2003)	x		x
Fish	+ (trout, Barton et al. (1985)					x
Mammals		= (bats, Lloyd et al. (2006)			x	
Herpetofauna	+ salamanders, Johnston and Frid (2002)	= (salamanders, Johnston and Frid (2002)	+ (salamanders, Crawford and Semlitsch (2007) + (toad, Goates et al. (2007) ? (Fr.w. turtles Burke and Gibbons (1995) ? (a snake, Roth (2005) + (herps, Semlitsch and Bodie (2003)		x	

+ (=, -) means that application of certification or a good management practice resulted in a positive (insignificant; negative) effect on biodiversity compared with the reference.

7.4 Green tree retention in clear-cuts

In clear-cuts, by definition, the original forest habitat and forest structure are totally lost. To preserve some of these habitats and their dependent species, standing trees can be left in the forest during the harvesting process. This biodiversity-maintaining practice is called green tree retention (GTR). In some cases it is anticipated that at least some of the retained trees will die to provide decaying and dead wood to species dependent on these resources. Also, so-called 'legacy trees' – those that have achieved near-maximum size and age – can be retained to provide specific habitats such as hollow stems. Retention can be as isolated trees or trees in clusters in set-asides.

Trees that are left standing during the logging process, are supposed to conserve some specific habitat features of the pre-logged forest. Species dependent on these features are thought to benefit from this habitat preservation. Expectations are that retention trees within clear cuts support higher numbers of species and individuals living in these habitats or species dependent on them and thus show a benefit to clear cuts without retention trees. Moreover, a positive relation is expected between the number of retention trees and biodiversity.

In the literature survey, the assumption was tested that forest management units where trees were retained contained a higher level of biodiversity than forest management units which were totally clear-cut, and that the level of biodiversity depended on the number of trees retained.

7.4.1 Tropical zone

No literature was found, presumably because clear-cutting is rare in tropical forests, at least as a forest management practice. Where tropical forest clear-cutting is practiced, such as perhaps in the pine forests of Central America, we found no studies on green tree retention.

7.4.2 Boreal zone

Hyvärinen *et al.* (2006) tested the effect of green tree retention (and controlled burns) on red-listed and rare deadwood-dependent (saproxylic) beetles in Finnish boreal forests for 2 years post-harvest. Sites with different densities of retained trees were monitored before and after logging. Some of the sites were burned one year after logging. Species richness was higher in the burned than the unburned plots, and increased with the density of retained green trees. The richness of red-listed and rare saproxylic species increased in the first post-treatment year, evidently due to the treatments, continued to increase on the burned plots in the second post-treatment year, but decreased in the unburned plots. The results showed that the living conditions of many red-listed and rare saproxylic species could be improved significantly with rather simple alterations to forest management methods. In particular, controlled burning with high levels of green-tree retention creates conditions favorable for many saproxylic species, but increasing the levels of green-tree retention in unburned areas can also be beneficial.

Toivanen and Kotiaho (2007) performed similar research on beetles in Finland, 1 to 16 years post-harvest. They also found that the abundance and species richness of saproxylic beetles were positively affected by burning, but the effect depended on tree retention. The difference between burned and unburned sites increased with the number of retention trees, and the effect of burning was not significant when there were fewer than approximately 15 retention trees/ha. Most important, the species groups that were unlikely to persist in conventionally managed forests (rare saproxylic and red-listed beetles), benefited substantially from burning and tree retention. The species richness of saproxylic beetles decreased with time since logging at both burned and at unburned sites. The authors concluded that burning of logged sites and leaving an adequate number of retention trees may be useful in the conservation of disturbance-adapted species (not desirable within our definition of biodiversity because they do not belong to the original species present) and can be used to improve the environmental quality of the matrix surrounding protected areas. Unfortunately, sites remained high-quality habitat for only a short time, thus spatial and temporal continua of burned areas must be ensured.

Lindhe and Lindelöv (2004) found that stumps of different tree species (spruce, birch, aspen and oak) provide breeding substrates for different species of saproxylic beetles in Sweden. Also the proportions of red-listed species differed per tree species. To optimize benefits the authors suggest that stumps should be located in sun-exposed logging sites and the focus should be on the tree species that provide substrate for the most threatened beetle species.

Lance and Phinney (2001) examined the effect of partial retention timber harvesting (15-22% not harvested) on birds in Canada, 2-5 years after logging. The undisturbed forest and the partial retention sites had similar numbers of species and similar numbers of individuals in all three years after logging. Not surprisingly, clear-cut forests had fewer species and individual birds. Species composition changed over time, with species being present in the partial retention sites that were not present in the undisturbed forest or clear-cut, and with some forest-dwelling species missing. These findings suggest that retention harvesting can serve to maintain most of the original bird species community.

Lõhmus *et al.* (2006) tested the effect of retention of isolated trees of different species on the conservation of bryophytes and lichens in Estonia. Bryophytes were reportedly very unhealthy throughout the retention patches, as indicated by the apparent desiccation of their shoots. Lichens were more robust, especially on some retention tree species (aspen and ash). Aspen hosted many more species than birch, including those of conservation concern. If tree species, size, and bark texture are carefully considered, green tree retention could be a successful treatment to promote lichen conservation. However, retention of single trees does not appear to provide sufficient protection for bryophytes, at least in the short term. Retaining trees in groups may increase their health but further research on this issue is needed.

Haeussler *et al.* (2007) studied “natural dynamics-based silviculture” in aspen-dominated boreal forests in Canada and compared the results to conventional clear-cutting. They used 12 plant community and plant functional group indicators to assess the effects of fire and partial cutting on biodiversity. Clear-cuts that were burned resembled areas affected by wildfires by reducing tall shrub abundance and regenerating post-fire specialists, but snags were lacking. The dual disturbance also retarded aspen re-growth and resulted in a 7-fold increase in non-native plants. In contrast, partial clear-cuts retained most attributes of uncut stands but after 3 years showed little evidence of accelerating development of mature stand characteristics. Thus, even though these silvicultural treatments resulted in and more natural variability in structure, composition and diversity than clear-cuts, further improvements in burn prescriptions and snag and green tree retention seem warranted.

Sullivan *et al.* (2008) studied the influence of variable retention harvests on plants and mammals 5-8 years after harvesting a coniferous forest in Canada. They studied clear-cuts, single seed-tree selection, group selection with seed-tree retention, patch cuts, and uncut forest sites. Species richness of herbs, shrubs, and total vascular plants were similar among harvested sites. Despite having no effect on total species richness of plants, retention of live trees provided suitable conditions for some plant species considered to be associated with forest interior conditions. Some of these species were present in retention harvest sites but not in clear-cut sites. Abundance, species richness, and species diversity of small mammals were maintained on all harvested sites, primarily because of habitat generalist and early successional species. The responses to the treatments were very species specific, and hence the authors recommended that a range of different harvesting systems should be used to maintain plant and mammal diversity across forest landscapes. These overall species responses highlight the need to evaluate small mammals on an individual and species-habitat association basis.

Koskela *et al.* (2007) studied the effect of green tree retention and rotation age on biodiversity conservation in commercial boreal forest in Finland. The decaying and dead wood created by retention trees and increased rotation age are supposed to promote biodiversity of old-growth forest species. They found through simulations that at the socially optimal choice, in which the amount of wood extracted from the forest and biodiversity conservation are both considered, the rotation age suggested by the current certification system (FSC) is not far from the optimal. However, the number of retention trees should be a lot higher than suggested by the certification system (5-15 times depending on what is defined as optimum) if rotation age and retention tree density were determined solely on the basis of maximized biodiversity conservation.

Matveinen-Huju *et al.* (2006) studied whether retention-tree groups in clear-cuts provided “life-boats” for spiders and carabid beetles in Finland, 3 years after logging. Their analysis revealed that the mean numbers of individuals increased with time-since-logging for many species of (semi-) open and/or dry habitat but decreased with time for many species requiring moist and/or wet forest habitat. Although

this study did not provide strong support for the importance of retention-tree groups in acting as 'life-boats' for the original forest species, their role in providing a continuum of coarse woody debris (CWD), large old trees, and other structural features in the managed forests is probably important, but in any case they should be carefully placed in the landscape.

Chan-McLeod and Moy (2007) studied whether residual tree patches could serve as stepping stones and short-term refugia for red-legged frogs in Canada. They found that residual tree patches can be important short-term refuges, but their value is size-dependent. Virtually all frogs released at the base of individual trees or inside small tree clusters left before the end of the 72-hour trial period but the proportion that left decreased curvilinear with increasing patch size. The harvest strategy for increasing the permeability of cut blocks to amphibians must balance the trade-off between patch size and distance between patches. Numerous small tree patches reduce inter-patch distances but are less likely than bigger patches to attract amphibians or be used as a stopover as frogs move through the harvested matrix. Larger tree patches at the same overall retention level are more effective in attracting close-by amphibians and in providing temporary habitat, but they may be spaced too far apart to be systematically intercepted by amphibians moving through the matrix. Frogs were less likely to leave tree patches with a running stream or where neighborhood stream density was high; clearly, whenever possible, retention patches should include streams or other wetlands.

Djupström *et al.* (2008) studied if set-asides in forest management units contribute to biodiversity in terms of dead wood volume and saproxylic beetles in Sweden. They explored three categories of set-asides: nature reserves, woodland key habitats (WKHs), and retention patches (i.e., groups of living trees left in clear-cut areas) and compared them among themselves and with old managed forest. No significant differences were found among the set-aside categories in terms of dead wood volume. Woodland key habitats had significantly more beetle species than retention patches and old managed forests as well as more red-listed species than retention patches. Saproxylic beetle species composition in retention patches differed from that of old managed forests and reserves. Despite differences in conservation values of these set-asides, the authors concluded that they provide valuable habitats and contribute (differently) to the preservation of old forest and saproxylic beetles.

Bradbury (2004) studied the influence of residual patch size on understory plant communities in aspen and pine-black spruce boreal forests in Canada. In both study areas, significant differences between understory plant communities in the pre-harvest forests and those in large, medium-sized, and small residual patches were observed. Not surprisingly, plant communities in residual patches were found to be intermediate between plant communities in pre-harvest forest and those in the clear-cut parts of the forest. One of the goals in leaving residual patches in clear-cut areas is to conserve the biodiversity found in pre-harvest forest so as to provide a source of propagules for plants not well adapted to colonizing disturbed areas. However, the ability of residual patches to serve as lifeboats for understory plant communities is influenced by several factors, among which is patch size.

The authors concluded that if forest managers expect to maintain pre-harvest understory plant communities within cut areas, residual patches need to be larger than those tested.

7.4.3 *Temperate zone*

Rosenvald and Löhmus (2008) reviewed 214 North American and European studies on green tree retention to determine whether, and under what circumstances, this silvicultural treatment meets its conservation objectives of:

- 'Life-boating' species over the regeneration phase;
- providing microhabitats for old-forest species in re-established forest stands and for disturbance-phase species on the recent cuts; and,
- enhancing species' dispersal by increasing landscape connectivity.

Compared with clear-cutting, green tree retention lowered the harvest-related population losses in 72% of studies, and nearly always improved the habitat for disturbance-phase insects and birds on the clear-cuts and for forest species in the regenerated stand. Green tree retention appeared to improve the habitat to different extents, and by different pathways, for different taxa. Among the species groups included in their meta-analysis, only the cover of grasses and herbs tended to be somewhat lower in the presence of trees. Other indicators of negative effects of green tree retention were the decreased abundances of some open-land birds, rodents, and regeneration of light-demanding tree species, but green tree retention seems to be flexible enough to provide openings (e.g., between tree-groups) for such species if desired. However, even though the optimum density and spatial configuration of retention trees have been listed among the most important questions for retention-harvest research, no general answer has been provided to date. Though grouping of trees seems to benefit a larger number of taxa, dispersed retention may be more appropriate for dispersal and for certain species groups. Another crucial point regarding the selection of retention trees is that characteristics other than species identity (such as age, size or shape of the crown) have received comparatively little attention from researchers.

Mazurek and Zielinski (2004) studied the benefit of 'legacy' trees (single retention trees that have achieved near-maximum size and age, which are significantly larger and older than the average trees in the landscape) on vertebrates (bats, birds, small and large mammals) in commercial forests in the USA. Randomly selected commercially-mature trees were selected as controls. As measured by species richness, species diversity, and use by a number of different taxa, legacy trees appear to add substantial habitat value to the studied forests, especially for nesting, roosting, and resting by bats and birds. This value is probably related to the structural complexity offered by legacy trees. The presence of a basal hollow was the feature that appeared to add the greatest habitat value to them and, therefore, to commercial forest stands. The results of this study call for an appreciation for particular individual trees as habitat for wildlife in managed stands.

Smith *et al.* (2008) studied the effect of tree retention on understory plant species. This treatment is intended to maintain or enhance structural complexity as compared to conventional silvicultural systems. They performed their study in hardwood-conifer forests in the USA, 4 years after treatment. The alternative systems had a positive effect: overall species composition and diversity were maintained. Techniques that enhance stand structural complexity may increase microsite variability on the forest floor and, as a result, sustain higher levels of understory plant diversity. By retaining biological legacies in the form of undisturbed patches of forest canopy and forest floor, these techniques may preserve late-successional species and those that are slow to recolonize disturbed sites. Despite overall maintenance of diversity under the treatment regime studied, the authors observed local extirpations of certain plant species, predominantly those with an affinity for late-successional habitats but noted that these extirpations also occurred in the control areas. Their results suggest that the magnitude and spatial pattern of retention are important for retaining understory species through a harvest cycle. They emphasized the need for more informed predictions of compositional changes in understory plant communities following different types and intensities of silvicultural disturbances.

In most plantations, spatial heterogeneity is deliberately reduced to promote the establishment and growth of planted tree species. To varying extents, biodiversity is viewed by plantation managers as “weeds and pests” at least insofar as other species reduce the productivity of the crop trees. In any case, plantations are less structurally and compositionally complex (i.e., they are typically monolayered and monospecific) than old growth natural forests. Tree retention is regarded as a key practice in creating complexity among them, providing heterogeneity in resources and in habitats. Yoshida *et al.* (2005) studied the effects of retention trees in a 60-year-old plantation in central Japan. Near the retained trees, the planted tree (*Larix kaempferi*) showed a reduction of 40 % to 60 % in basal area, presumably due to the shading effect. In contrast, the spatial gradient of shade and colonization opportunity provided by retained trees greatly affect the distribution of the colonized species, according to their shade tolerance and seed dispersal ability, which resulted in the stand structure with a heterogeneous shrub-layer vegetation. Thus retention proved particularly important for the enhancement and long-term maintenance of structural and compositional complexity in these plantations. However, in considering the merchantable production of planted trees, the considerable loss in basal area in the nearby area must be taken into account. A possible solution may include the use of more shade-tolerant species as planted trees, and the thinning of colonized neighbors, so as to reduce competition in the nearby area.

7.4.4 Summary

No studies on tree retention in the tropics were found. This is no surprise since tree retention is a potentially biodiversity-conserving treatment where clear-cutting is used for timber stand management, which is common in boreal and temperate zones only. The findings are summarized in table 12.

Solitary green tree retention has been reported to have a positive effect on preservation on species of birds (Lance and Phinney 2001), beetles (Hyvarinen *et al.* 2006; Toivanen and Kotiaho 2007), and wildlife in general (Mazurek and Zielinski 2004; Rosenvald and Löhmus 2008). Birds generally benefit from tree retention with respect to numbers of species and individuals, but one study reported that bird community composition differed from undisturbed forest even where green tree retention was practiced (Lance and Phinney 2001). As far as saproxylic beetles are concerned, there is a positive relation between the number of retention trees and species conservation, especially the rare and red-listed taxa (Hyvarinen *et al.* 2006; Lindhe and Lindelöw 2004; Toivanen and Kotiaho 2007). In some forest types, fires augment the benefits of green tree retention and are necessary in the long term to ensure the presence of appropriate forest beetle habitats (Hyvarinen *et al.* 2006; Toivanen and Kotiaho 2007). Some research has also shown that the types of trees that is retained affects beetle populations (Lindhe and Lindelöw 2004).

In regards to plant conservation, green tree retention does not seem to have a large impact (Haeussler *et al.* 2007; Sullivan *et al.* 2008). In contrast, lichens reportedly grow well on retained trees. As for beetles, the tree species retained influences the number of lichen species retained in clear-cut areas (Lohmus *et al.* 2006).

One study in Finland reported that the currently required density of retained trees is not sufficient for the purposes of biodiversity conservation (Koskela *et al.* 2007). Other studies also report that current rules related to the numbers and sizes of retained trees are not sufficient to maintain populations of certain species of concern (Haeussler *et al.* 2007; Lohmus *et al.* 2006). Given that there are often species specific responses to tree retention (Sullivan *et al.* 2008), more research is apparently needed (Lohmus *et al.* 2006; Rosenvald and Löhmus 2008).

Tree retention in groups reportedly has positive conservation benefits for plants (Bradbury 2004; Smith *et al.* 2008) and frogs (Chan-McLeod and Moy 2007). A study on beetles (Djupström *et al.* 2008), reported that although there were species richness benefits of green tree retention, the beetle species composition changed when compared to old growth forest. For spiders and carabid beetles, in contrast, green tree retention reportedly does not have a marked effect compared to clear-cuts (Matveinen-Huju *et al.* 2006). All authors mention that in order for group tree retention to be effective, the sizes of trees and distances between patches should be carefully considered as well as patch quality. For instance, frogs showed a clear preference for large patches (Chan-McLeod and Moy 2007) and for plant communities, size is an important factor (Bradbury 2004; Smith *et al.* 2008).

We found only one study on green tree retention that was carried out in a plantation. Retention trees reportedly reduced the basal area of planted trees, but increased the heterogeneity (species composition) of the shrub layer (Yoshida *et al.* 2005).

Overall, tree retention seems to benefit both plants and animals, but retained tree species, densities, locations, and the sizes and connectivity of patches all influence the magnitude of the conservation contribution of green tree retention.

The table below summarizes the main trends in the responses found in the literature, against two hypotheses. Because of the nature of this management practice – retaining (groups of) trees in landscapes which are largely clear-cut – there will always a large effect of logging on total abundance of species, so this aspect is not taken into consideration in the table.

Table 12. Summary of the effects of Green Tree Retention on different taxa.

Green Tree Retention	Comparison of forest management units where trees are retained with forest management units without	Comparison of forest management units where trees are retained with undisturbed forest landscape	Tropical	Temperate	Boreal
Biodiversity	+ Rosenvald and Löhmus	- (Koskela et al. 2007)		x	
Birds	+ (Lance and Phinney 2001)	= (Lance and Phinney 2001)			x
Invertebrates	+ non-forest (!) beetles (Hyvarinen et al. 2006; Toivanen and Kotiaho 2007) ? spiders, beetles Matveinen-Huju et al. (2006) +/- beetles (Djupström et al. 2008)	+/- beetles (Djupström et al. 2008)			x
Lower plants	? bryophytes Löhmus et al. (2006) + lichens Löhmus et al. (2006)				x
Plants	= (Haeussler et al. 2007; Sullivan et al. 2008) + Bradbury (2004) + Smith et al. (2008)	- Bradbury (2004)		x	x
Mammals	=/? Sullivan et al. 2008 + Mazurek and Zielinski (2004) (legacy trees)			x	x
Herpetofauna	+ frogs (Chan-McLeod and Moy (2007)				x

+ (=, -) means that application of certification or a good management practice resulted in a positive (insignificant; negative) effect on biodiversity compared with the reference.

7.5 Protected areas within forest management units

Protected areas within production forests (=reserves or set-asides) consist of areas within production forests where no resource extraction is permitted. The justification of retaining these reserves within production forests is widely accepted insofar as they are expected to provide refuges for wildlife sensitive to logging, protect critical wildlife habitat (e.g., breeding and feeding areas), and protect areas that are particularly sensitive to logging damage (e.g., water sources and wetlands, steep slopes, riparian zones and soils with especially low trafficability). In addition, ecological reserves serve as a legacy of undisturbed forest for the purpose of monitoring logging impacts and have heritage values as examples of undisturbed ecosystems. Generally accepted criteria for demarcation of protected areas include key habitats for priority wildlife species, buffer areas around streams, scarce and declining habitats, and specialized habitats. Priority species include those that are threatened, endangered, or of special concern for other reasons. They also include regional or local endemic species. Their habitats may include nest or den sites, resting sites, or important feeding sites. In other cases, these areas are defined as

representative samples of normal production forest (Fredericksen and Pena-Claros 2007; Frumhoff 1995).

Protected areas contribute to the maintenance of biodiversity in certified forests if it can be demonstrated that forest management units with protected areas contain higher levels of biodiversity than those without. We believe that their effectiveness is strongly dependent on size, shape, quality, location and connectivity (Fredericksen and Pena-Claros 2007; Frumhoff 1995; Sayer *et al.* 1995), particularly in forests heavily altered by logging. The more different the landscape surrounding protected areas (ranging from well-managed production forest to forest plantations and clear-cuts), the more important their function and the less likely they are to maintain high levels of forest biodiversity. Therefore we expect variable outcomes with respect to their ability to conserve biodiversity.

7.5.1 Tropical zone

Barbosa de Lima *et al.* (2008) compared 7 FSC certified and 7 non-certified Eucalypt and Pinus plantations in Southern Brazil on a number of conservation parameters including the protection of Permanent Preservation Areas and their surroundings. No direct impacts of certification on biodiversity were reported, but it was demonstrated that certification had an impact on the management of areas in the proximity of these areas: the demarcation of sensitive natural areas, pre and post-harvest evaluations in buffer zones, direction of harvest logging, and the identification of trees for the conservation of avifauna.

Laidlaw (2000) studied species richness and composition of mammal communities in a network of small unlogged forested areas in Malaysia. These areas are protected, unlogged, and usually embedded in a mosaic of logged forest. The size of the remaining area of natural forest in which the protected area was located largely determined the species richness and composition in the protected area. For instance, when areas of natural forest of ≤ 459 ha were considered, size was critical: a sharp loss in mammal species richness was apparent between 70 and 164 ha. Differences in habitat quality accounted for the differences between protected areas and the adjoining vegetation. Even though small undisturbed areas can effectively enhance the mammal communities in disturbed landscapes, the survival of some mammal species, including the largest carnivores and herbivores, will be determined by the total accessible area of protected forest or at least forest in which hunting is effectively controlled.

7.5.2 Boreal zone

In conservation research carried out in boreal forests, the phrase “woodland key habitat” (WKH) is often used. WKH refers to ‘especially valuable habitats from a conservation point of view, where red-listed species are expected to be found’ (Sverdrup-Thygeson 2002).

In Fennoscandia, the concept of WKHs has become one of the key concepts for defining hotspots for forest biodiversity. Nevertheless, even in the relatively well studied forests of that region, there is a serious lack of research on the diversity value

of the assumed key habitats to many important groups of organisms. Junninen and Kouki (2006) studied whether WKHs in Finland were hotspots for wood-decaying fungi. The large total number of species they found in WKHs supports the underlying assumption that, compared with the surrounding matrix, WKHs are richness hotspots for these fungi. However, the number of red-listed species they found in the WKHs was very small. Thus, it can be concluded that, compared with production forests, WKHs can maintain a rich fungal flora, but they may contribute little to the conservation of some threatened taxa.

Sverdrup-Thygeson (2002) investigated the forest history, structural characteristics, and the species composition of saproxylic beetles in 30 old-growth forest WKHs, and compared them with production forest of the same age in Norway. No statistically significant differences in forest characteristics, saproxylic beetle communities, or number of red-listed beetles were found between the WKHs and the production forest because, the authors argued, of a combination of logging in the WKHs prior to their being declared as such (signs of logging practices were present), and insufficient size of the WKHs. The authors conclude that it is important to protect the few remnants of old growth and at the same time restore old-growth conditions in other stands in heavily exploited landscapes to maintain the species dependent on old-growth forest and to fulfill the requirements of sustainable forest management.

The almost complete absence of the rarest species in WKHs reported in several studies may be related to the amount and quality of dead wood and/or to habitat fragmentation. Old trees and fallen logs have been found to be crucial habitat components for the occurrence of red-listed fungi and bryophytes in woodland key habitats in Sweden (Berg *et al.* 2002). These aspects, however, are not considered in the definition of WKHs in Finland, which may be why their role in conservation of wood-decaying fungi was limited. It is possible that the quality of the Finnish key habitat sites will improve in this respect as time passes. However, the small size of the key habitat stands and their location in fragmented landscapes may override the positive development. Fragmentation and isolation may have already contributed to local extinctions of the rarest species.

7.5.3 Temperate zone

No studies found

7.5.4 Summary

Protected areas and woodland key habitats (WKHs) in forest management units have been understudied with respect to their effect on biodiversity conservation (Table 13). The review by Bruner (2001) shows that parks in the tropics can conserve biodiversity, especially if hunting is controlled. It seems reasonable to extend this sort of result to protected areas within forest management units, but there are few data in support of such an extension. One tropical study showed that size and connectivity of protected areas affects their conservation success (Laidlaw 2000). Researchers also recommend that WKHs in temperate and boreal regions should also be 'sufficiently large' (Sverdrup-Thygeson 2002) and well-connected (Berg *et*

al. 2002) if they are to contribute to the conservation of many threatened and rare species (Junninen and Kouki 2006; Sverdrup-Thygeson 2002).

Table 13. Summary of the effects of protected areas within forest management units on different taxa.

Protected areas	Comparison of forest management units including protected areas with forest management units without	Comparison of protected areas in managed forests with undisturbed forest landscape	Tropical	Temperate	Boreal
Mammals	+ Laidlaw (2000)	- Laidlaw (2000)	x		
Invertebrates	= beetles (Sverdrup-Thygeson (2002)				x
Fungi	+ Junninen and Kouki (2006)				x

+ (=, -) means that application of certification or a good management practice resulted in a positive (insignificant; negative) effect on biodiversity compared with the reference.

7.6 High Conservation Value Forests

The term High Conservation Value Forests (HCVFs) refers to forests of outstanding and critical importance. By definition, this outstanding or critical importance is due to the presence of an unusually high number of rare species (plants or animals), and/or to the critical importance of the forest to local people because it provides them with food or income when they have few alternative sources, and/or to its cultural importance (FSC 2008). Unlike the other forest management tools discussed in this report, HCVF are exclusively associated with certified forests, owing their existence to the formulation of FSC Principle 9.

The reason why HCVFs are treated differently in certification standards than 'ordinary forests' is that HCVFs are not just demarcated on the basis of the presence of one or two rare species. Instead, HCVFs are selected because of a concentration of rare species or due to a level of social importance that renders the area much more important than 'ordinary' forests in the area. The forest managers responsible for HCVFs have a level of responsibility above and beyond what would be expected in 'ordinary' forest management (as explicitly recognized in FSC Principle 9) (FSC 2008).

For example, to comply with the FSC requirements in a HCVF (Principle 9, FSC), forest managers need to demonstrate that:

- They have identified which of their forest might be considered 'High Conservation Value Forests';
- they have done this in consultation with other people who might have a relevant opinion;
- they have made sure that the way they use and manage the forest does not negatively affect the critical values they found; their management practices can affect the forest, but not the critical 'values' or 'critical qualities' on which the HCVF was selected;

- they have a system for assuring that the critical values or qualities are being protected.

Overall, forest managers are required to adapt their management plans in HCVPs so as to avoid reducing their special values (FSC 2008).

The extent to which biodiversity can be conserved by HCVPs will depend on which area is defined as HCVP and how it is managed. In contrast to protected areas, logging does occur in HCVPs. In that respect, they may resemble forests with extreme forms of RIL applied. We therefore expect that HCVPs perform better for conserving biodiversity than areas not managed as HCVP, while they would perform not as good as fully protected areas. However, logging happens in a context of elevated biodiversity values, so the impacts may also be expected to be larger.

Both certifiers and forest managers have struggled with the concept of HCVP, and in the history of forest certification it has not yet established as a clear forest management tool. As a result, studies on the performance of HCVP are rare and none were found dealing with biodiversity.

7.7 Corridors

Corridors are defined as linear habitats, embedded in a dissimilar matrix, that connect two or more larger blocks of habitat. Habitat corridors are established for conservation purposes on the grounds that they enhance or maintain the viability of populations of species of concern (Beier and Noss 1998). Creating and restoring corridors between isolated habitat patches can help mitigate or reverse the impacts of fragmentation (Williams and Snyder 2005). In addition to acting as travel pathways, corridors may provide the basic requirements for foraging and breeding (Perault and Lomolino 2000 and references therein). Establishing corridors between unlogged patches in logged forests, for example, is thought to reduce biodiversity loss by protecting specific habitats and their biodiversity and by providing connectivity between habitats that serve as refuges for plants and animals that are sensitive to logging. While this sort of corridor is the most relevant for our evaluation of the biodiversity benefits of forest certification, we review the literature more broadly.

Falcy and Estades (2007) performed model simulations to compare the effectiveness of corridors relative to enlargement of habitat patches. Their results indicate that, for a given amount of habitat, patch enlargement increases population size on average more than the establishment of biological corridors. Enlarging patches is most effective if it causes them to surpass the minimum size required to maintain viable populations or if their degree of isolation from other patches substantially diminishes the usefulness of corridors as movement conduits. Knowledge of a species' density–area relationship, the minimum patch size required for maintaining a viable population, and the degree of isolation between patches should inform decisions about whether to establish a corridor or to enlarge a habitat patch. The authors indicate that while corridors may mitigate potential effects of inbreeding depression, they are not always the best method for conserving fragmented populations.

Because of their nature, riparian buffers along rivers and streams can be considered corridors as well, but they were treated in section 7.3.

7.7.1 Tropical zone

Metzger (1997) studied the relation between landscape structure and tree species diversity in a fragmented forest in Brazil. Forest connectivity provided by forest corridors and matrix stepping stones apparently helped maintain tree species diversity. The species composition of the forest fragments appears to be related to the spatial arrangement of neighboring forest patches, and is also affected by matrix complexity. These results suggest that the spatial arrangement of forest patches and the complexity of the matrix may be more important to the survival of the studied species than fragment size and degree of isolation.

Tropical plantations represent a rapidly expanding source of industrial wood. In Indonesia, such large-scale industrial plantations generally consist of large mono-specific blocks interspersed with natural forest remnants. The extent and biodiversity value of these remnants vary as laws and regulations on their design and management are either unclear, without solid scientific basis, or left to the interpretation of private companies responsible for the plantations. Nasi *et al.* (2008) studied the impact of landscape and corridor design on primates in an industrial plantation landscape. They found that primates were not abundant in plantations but were present in forest remnants that were connected to larger areas of natural forest. Habitat quality in terms of crown closure also played a major role in primate abundance. The authors concluded that these remnant natural forests may, if appropriately designed and managed, be used to mitigate the negative impact of plantations on biodiversity especially if they are designed so as to maintain some degree of connectivity with and between remaining natural forest patches.

Anzures-Dadda and Manson (2007) studied landscape-scale effects on howler monkey distribution and abundance in 119 rainforest fragments in Mexico. Monkey abundance increased in forest fragments with a higher number of corridors creating links with other fragments. The authors reasoned that by using vegetation corridors to travel among forest fragments that exhibit a range of conditions in resource quantity and quality, monkey troops should be better able to meet their needs and this should be reflected in greater overall population sizes. More studies are needed to clarify our understanding of the role of vegetation corridors in the movements and metapopulation dynamics of wildlife in fragmented landscapes.

Parren *et al.* (2002) studied usage of shelterbelts established in the mid-1930s in Ghana as wildlife corridors for forest elephants. Shelterbelt reserves are strips of forest often not more than 1.5 km wide and up to 20 km long. Three out of four observed shelterbelts showed evidence of elephant passage. This demonstrates that corridors could be successfully used to connect existing national parks and forest reserves into a forest network area, although it is not clear whether elephants persist thanks to these corridors.

7.7.2 Boreal zone

No studies found.

7.7.3 Temperate zone

Kondo and Nakagoshi (2002) studied the effect of forest structure (vegetation diversity) and connectivity on bird distributions in a riparian landscape in Japan. The study area consisted of 33 small forest fragments around two rivers; this linear cluster of small forests connected two large forests. Birds selected suitable forests in a linear cluster for movement between two large forests and for foraging according to their biological attributes (forest interior and forest edge birds). Most birds did not use the small forests as 'stepping stones' where they could stop temporarily while moving by the shortest route between the large forests; instead, they used the linear cluster of small forests as a corridor. The linear cluster of small forests facilitated bird movement across an otherwise inhospitable landscape toward isolated habitat remnants.

Woodland isolation by removal of hedgerows in an effort to consolidate land is recognized as a major threat to biodiversity in Europe. In one study on this topic, Pereboom *et al.* (2008) evaluated corridor use by pine martens in a European fragmented landscape. Martens were not confined to large forests. Instead, they made substantial use of hedgerows, especially for foraging. They stayed close to forest cover when venturing into open fields, indicating a certain dependence on hedgerows.

Damschen *et al.* (2006) performed a large-scale study in the USA on the effects of corridors on plant species richness. The most parsimonious explanation for the increased plant species richness they observed in connected patches is that corridors alter the balance between three important processes and interactions in ways that promote diversity. Corridors promote: (a) colonization by increasing seed deposition; (b) promote within-patch recruitment by increasing pollen movement; and (c) alter foraging by seed predators that could benefit species otherwise likely to suffer intense seedling competition. Although individual plant species and their interactions differ in their responses to corridors, the results show that for 300 plant species, connecting patches with corridors has positive net effects on native plant species richness. By providing experimental evidence that corridors increase the number of native plant species in large-scale communities over a wide range of environmental conditions, the authors show that corridors are not simply an intuitive conservation paradigm, but a practical tool for preservation.

Haddad *et al.* (2003) studied corridor use by 10 different species, including butterflies, small mammals, and plants in the USA. Corridors allowed more movements between connected than unconnected patches for all species, indicating that movements of disparate taxa with broadly different life history traits and functional roles benefit from corridors.

Tewksbury *et al.* (2002) reported the results of a large-scale experiment to test the effects of corridors on plants, animals and their interactions in the USA. Their

replicated experimental design applied in a pine plantation in South Carolina, consisted of clearcut patches of the same size but of different shape and degree of connectedness. Butterfly and birds exchange between patches was stimulated by corridors, which facilitated two key animal-plant interactions, pollination and seed dispersal. Fruit set (a function of pollen movement) and seed dispersal were higher in connected than unconnected patches. The authors argue that increased fruit set and seed movement between connected patches will have additive effects on gene flow and population dynamics. Given that plants producing more fruits are likely to attract more frugivores, plants in connected patches are likely to contribute more to gene flow both within and between patches due to increases in pollen movement, fruit removal, and seed movement down corridors. These results show that the beneficial effects of corridors are more than simply the effect of increased habitat area.

Perault and Lomolino (2000) studied the influence of corridors on mammal community structure across a fragmented ecosystem in the USA. Habitat characteristics and species assemblages differed significantly among the corridors studied. The adjacent habitat played a large role in determining the number of species found in a corridor: less old-growth forest surrounding a corridor resulted in fewer forest-dependent species.

Beier and Noss (1998) reviewed 32 studies that addressed whether corridors enhance or reduce the population viability of species in habitat patches. Although not made explicit, most studies were apparently conducted in temperate zones. Evidence from well-designed studies generally supports the utility of corridors as a conservation tool. Almost all studies reviewed show that they provide benefits for and are used by animals in the landscape but it remains difficult to make generalizations about the effect of corridors because of the species-specific responses. Corridor effectiveness also varies with landscape characteristics.

Another review by MacDonald (2003) found that designing and assessing the benefits of corridors for biodiversity conservation is difficult because of the large number of potential influences. Use of corridors by fauna may vary with vegetation type, corridor dimensions, and geographical location. Use of corridors varies markedly between species, even taxonomically closely related species, and can even vary intraspecifically with gender or age. The conservation value of corridors for plants has been addressed by relatively few studies. Nevertheless, there is strong evidence to suggest that corridors are an effective supplementary conservation measure—they must accompany other conservation solutions such as reservation of extensive forested areas, manipulation of fire regimes, and specific logging prescriptions. The authors conclude that there are still wide gaps in our understanding of the role of corridors.

7.7.4 Summary

All studies on corridors we reviewed, including two other similar reviews, indicate that corridors have positive effects on species and individuals (Table 14). Some of the studies reviewed focused on processes that may be expected to be beneficial for the

maintenance of biodiversity, such as migration (Pereboom *et al.* 2008, Haddad *et al.* 2003), pollination and seed dispersal (Tewksbury *et al.* 2002), rather than on direct biodiversity measures (such as diversity and composition). Yet, the conclusion of a positive effect appears to be valid for plants (Damschen *et al.* 2006; Haddad *et al.* 2003; Metzger 1997; Tewksbury *et al.* 2002), primates (Anzures-Dadda and Manson 2007; Nasi *et al.* 2008), birds (Kondo and Nakagoshi 2002; Tewksbury *et al.* 2002), several mammals (Haddad *et al.* 2003; Peereboom *et al.* 2008; Perault and Lomolino 2000), and butterflies (Haddad *et al.* 2003; Tewksbury *et al.* 2002). This overall conclusion notwithstanding, effectiveness of corridors varies a great deal (Perault and Lomolino 2000). Habitat connectivity and quality are the two primary variables that control the conduit function of corridors. Habitat connectivity is determined by corridor density, distance between patches, and the spatial arrangement of patches (Anzures-Dadda and Manson 2007; Kondo and Nakagoshi 2002; Metzger 1997). Habitat quality depends on the physical aspects of the landscape (for example, cover, type of vegetation, moisture, elevation) (Beier and Noss 1998; Nasi *et al.* 2008). While corridors are generally considered valuable, a modeling study noted that for a given amount of habitat, patch enlargement can have greater benefits for population size of species than the establishment of corridors (Falcy and Estades 2007).

According to MacDonald (2003) who reviewed corridor studies, there are still wide gaps in our understanding of the role of corridors. Since no corridor is the equivalent to another and because species show specific responses to corridors (Beier and Noss 1998; Damschen *et al.* 2006; MacDonald 2003), site and species-specific research is generally needed to determine the effectiveness of corridors for biodiversity conservation.

Table 14. Summary of the effects of corridors on different taxa, based on published studies.

Corridors	Comparison of forest landscapes with corridors between patches with forest landscapes without corridors	Comparison of corridor habitat with undisturbed forest landscape	Tropical	Temperate	Boreal
Biodiversity	+ Haddad <i>et al.</i> (2003), Beier and Noss (1998), MacDonald (2003)			x	
Mammals	+ monkeys Nasi <i>et al.</i> (2008), Anzures-Dadda and Manson (2007) +? elephants Parren <i>et al.</i> (2002)		x		
Birds	+ Kondo and Nakagoshi (2002) + Tewksbury <i>et al.</i> (2002)			x	
Mammals	+ Pereboom <i>et al.</i> (2008)			x	
Plants	+ Damschen <i>et al.</i> (2006)			x	
Invertebrates	+ butterflies Tewksbury <i>et al.</i> (2002)			x	

+ (=, -) means that application of certification or a good management practice resulted in a positive (insignificant; negative) effect on biodiversity compared with the reference.



8

Evaluation and discussion

This report reviews 67 studies on the effects of good management practices associated with forest management certification on aspects of forest biodiversity. Just a few studies directly compare aspects of biodiversity before and after certification of forest management. These reported positive effects of certified forest management on biodiversity, although data supporting the findings of these studies were not always presented. Other studies on the impacts of good management practices provided more information, but knowledge gaps remain. Not all practices were studied in all geographical regions, and not all studies were conducted with the intention of revealing the effects of the practice on biodiversity. Therefore, conclusions about the effects of certification on biodiversity are partly based on what we believe are reasonable predictions of the benefits of changes in forest management practices.

The main conclusion is that in spite of a very large variety in responses between species, **the forest management practices associated with forest certification appear to benefit biodiversity in managed forests.** This is further elaborated upon in Chapter 8.1. This is in agreement with information gathered through discussions with certifiers and forest managers, and with our own experiences in the field, and aligns well with the conclusions of reviews of the impact of certification by FSC (Karmann and Smith 2009). Despite the apparent differences among the certification systems in the rigor with which biodiversity concerns are addressed, the planning, supervision, and basic good management practices required by all serve to mitigate many of the deleterious environmental impacts of logging and other forest management activities.

This main conclusion must be qualified by a number of additional observations:

- There is a very high variation, both in forest management practices associated with certification and in responses between and even within species;
- There is little quantitative evidence about the long-term effects of certified forest management on biodiversity;
- There are few data on which to base the conclusion that certified forest management is sustainable in terms of biodiversity conservation at the level of populations and communities – we simply don't know;
- Well-managed (certified) forests are not equivalent to undisturbed forests in terms of conserving (forest) biodiversity.

A large number of provisos and considerations accompany these main conclusions, as outlined in the next few sections.

8.1 Good management practices and biodiversity

It is clear that negative effects of logging on forest species are reduced when applying RIL since it causes less damage to the forest than conventional logging techniques. Riparian zones offer specific habitat characteristics of which many aquatic, semi-aquatic, and terrestrial species are dependent for many, if not all, stages of their life cycles. Protecting these zones against logging damage thus contributes to species preservation although the extent to which species benefit from these protected zones depends on zone width and several other factors. Green tree retention in clear cuts maintains some of the habitats present before logging, on which many species depend. Thus, when compared to total clear cuts, retention trees provide a benefit to many species but the magnitude of the benefit depends on the type and the number of retention trees. Corridors provide shelter to many species and provide links between otherwise isolated patches of remaining habitat. Therefore, they benefit many species in intensively logged areas. Size, shape, and connectivity of these corridors determine their effectiveness in species conservation. It seems logical that protected areas within logging units and HCVF protect many species from the negative impacts of logging, but we found few data to support this conclusion.

The management activities we considered provide benefits to certain species and species groups and therefore are effective in conserving biodiversity to a certain extent. However, the effectiveness of any certification-required activity depends on the way it is implemented. We found no studies that described the combined effects of all these practices within one forest unit. Last, we should bear in mind that these measures may have a positive effect on biodiversity when compared to conventional logging, but often not when compared to undisturbed forests (see discussion on 'Acceptable change' in Chapter 8.3.3).

Having established (or at least argued with evidence) that certification benefits biodiversity, the next question is "by how much?" Any answer to this question obviously needs to be couched in abundant caveats; e.g., which forest, which certification system, which biodiversity components, what time scale, compared to what.

In forests with a well-established silvicultural tradition such as central Europe, it is likely that changes in management practices prompted by certification will be marginal and result in small effects on biodiversity. In forests managed in monocyclic rotational systems (clear cutting), management practices such as Green Tree Retention, maintenance of streamside buffers and establishment of corridors and protected areas may have a very significant effect on the conservation of forest species. Similarly, in tropical forests with selective but uncontrolled logging, management practices such as RIL, protection of buffer zones and protected areas could make a big difference for forest biodiversity. For these forests, we suggest that in global assessments of the biodiversity impacts of different land-use practices, certified forests should be differentiated from otherwise managed and exploited forests.

In spite of the substantial contribution that forest certification can make to the conservation of biodiversity, the evidence shows that there are also many species and ecosystems which are negatively affected by any form of logging. Therefore, sound conservation strategies must be grounded on an adequate mixture of protected areas and well managed forests.

8.2 Limitations of the study

8.2.1 Choice of management practices

We quite strictly examined the effect of certain forest management practices on biodiversity, regardless of the context in which logging occurs. Many other practices associated with forest certification were not examined, such as control of hunting, management of chemicals and oil, road construction, etc. The same holds for soil scarification, liana cutting, controlled burns, post-harvest liberation of potential crop trees and other silvicultural practices. In many tropical countries, logging, regardless of the standard applied, is only the first step in a chain of events that may lead to complete degradation of the forest, such as further (illegal) logging, slash and burn agriculture, invasion by humans, hunting, etc. If forest certification is successful in mitigating these deleterious secondary effects of logging, its (relative) effect on biodiversity will be quite positive, but this was not the focus of the present study.

In particular, increased hunting pressure on wildlife is considered one of the most severe secondary impacts of logging as far as biodiversity is concerned. The defaunation that results indirectly from the increased access provided by logging roads is a huge problem throughout the tropics (for reviews see Fimbel *et al.* 2001; Haworth and Counsell 1999; Peres *et al.* 2006; Robinson and Bennett 2000). Although all certification systems have specific prohibitions against unsustainable hunting, and although environmentalists have complained about the lack of attention to wildlife in certification audits (Bennett 2003), there are apparently no data for determining whether certification serves to reduce hunting pressure.

8.2.2 Data availability and quality

So far, a systematic effort to study and understand the effects of certified forest management on biodiversity conservation – a major rationale for certification – appears to be lacking, both from the side of certification agencies and their clients, forest managers, and from the side of research community. The study of biodiversity is complex, time consuming and expensive, and its results are hard to interpret. While this is so, this is not matched by formulating clear conservation objectives (and monitoring those) by forest management practitioners and auditors, nor by establishing clear research protocols by the research community interested in these issues. Exceptions exist, such permanent sample plot censuses for timber species growth and yield studies and long-term breeding bird censuses, but these have limitations in scope and/or geographic coverage. As a result, our understanding of certification effects is not expected to increase in another than haphazard manner in the near future.

For practical reasons, research considers the effects of management practices on just a few taxa, sometimes just one species. The variation in responses between species, and the small percentage of 'biodiversity' covered make generic conclusions about management effects on 'biodiversity', the full suite of species in the forest, speculative.

As mentioned, studies comparing biodiversity (i.e., species composition, richness, and abundance) before and after certification are scarce. Forest certification is a relatively new development, and studies comparing its long-term effects on biodiversity require substantial time, although it should be possible to detect some short-term responses for some species or species groups. Strange enough, studies on short-term impacts have apparently not yet been published.

Information is not equally distributed over biomes and forest management practices. The literature on forest management in boreal and temperate zones is much more extensive than in the tropics (even though we included approximately equal numbers of studies from each biome). While some management practices and management systems received ample attention from researchers, others were seldom studied. For instance, there are many studies on the impacts of (riparian) corridors and RIL, while fewer were found on the impact of HCVMs and just a few studies were found about plantations. Even when available, many studies about management practices were not designed to evaluate pre- and post-treatment biodiversity. This resulted in studies not mentioning whether or not the research was conducted in a certified forest, or in the lack of a control forest (which would ideally be a comparable, conventionally logged forest, instead of 'good management' logging). Some species groups are clearly more popular among researchers than others. For instance we could find many more data on birds than on earthworms. The review considered studies of all species we could find during our search, but for many other species groups information is lacking.

8.2.3 Availability of knowledge on biodiversity

It became apparent in the course of our study that the extent to which forest certification can be demonstrated to help maintain forest biodiversity varies with the availability of scientific knowledge of species and their reactions to different forest interventions. It also seems to vary with the size and activity levels of local and regional groups of environmentalists. But even in low diversity and well studied ecosystems where there is substantial pressure for conservation, much of the information needed to evaluate the impacts of forest management and certification is still lacking. The challenge for forest managers, certifiers, and biodiversity researchers in tropical forests are probably at least an order of magnitude larger. While some international environmental groups are critical of certification, those that accept that logging of some sort is likely to continue might consider contributing to the building of local capacity to identify some of these species and to learn about how to minimize the deleterious impacts of forest management on them.

The availability of information about the biodiversity in forest management units varies tremendously, especially between low diversity forests in temperate and boreal biomes, to hyper-diverse forests in the tropics. In the former, environmentalists may have access to data justifying concern about the impacts of forest management activities on rare species of lichens, mosses, and even fungi. In startling contrast, species of plants, birds, and even primates new to science are regularly discovered in tropical forest management areas. But while scientists may have names for most temperate plant, bird, and mammal species, they generally have little or no information about how the majority of these species will react to a variety of forest interventions. And as for other taxa, such as invertebrates and microbes, our biodiversity knowledge base in even the best known temperate forests is generally limited. For example, a team of researchers in the Pacific Northwest of the USA started in 1994 to develop the scientific basis for management practices that ensure the continued viability of populations of all species associated with old growth forests. In their list of 1100 species, 400 were defined as rare or little-known (Rafael and Molina 2007).

Researchers always conclude that more research is needed, but in the case of the biodiversity benefits of forest certification, this conclusion seems entirely warranted. Over the course of not yet 20 years, forest certification has done more to change tropical forestry than any of the many preceding and simultaneous interventions with similar intentions (e.g., the Tropical Forestry Action Plan, the Montreal Process, and the ITTO's many outstanding efforts). Unfortunately, despite cries for increased involvement of biologists (Bennett 2001; Putz and Viana 1996), pleas for increased attention from researchers (Putz 1996; Putz and Romero 2001), and the publication of numerous books on the topic (e.g., Upton and Bass 1996; Viana *et al.* 1996; Vogt *et al.* 2000), definitive studies on the biodiversity effects of forest certification remain to be conducted. Such a study might also address the other expected but as yet seldom-measured benefits of forest certification including improved worker safety, protected ecosystem functions, and increased future timber yields.

8.2.4 Researchers and their views

Researchers concerned about the biodiversity impacts of environmental interventions quite naturally focus on the immediate effects of the most severe treatments. Such is the case with studies on the biodiversity impacts of forest management. Few of the studies we reviewed considered forest conditions more than four years post-logging. In the few cases in which forest recuperation was monitored for longer periods of time, recovery during the first five years was substantial for processes such as rates of sediment loading of streams, forest biomass, and microclimatological conditions. Similarly, even in forests subjected to low-intensity selective logging in which logging roads and skid trails cover a small percentage of the soil surface, they are the focus of a large proportion of the environmental assessment research related to logging. When the focus of this research is on soil drainage and hydrology, this focus is entirely justified. In contrast, it is not reasonable to draw forest-wide conclusions about birds and other biodiversity components from studies focused on scattered sites that were severely disrupted.

Another limitation with the available research on the effects of different management practices is that many studies seem motivated by the desire to demonstrate that logging has impacts of forests so as to make a case for the cessation of logging (as observed above in section 7.2.1). This strategy has been effective in some areas, such as on government owned forest lands in the USA. Logging rates have been especially curtailed in the US National Forests, many of which are now essentially managed as protected and recreational lands. The same holds for forests held by the major land owners in the Netherlands, where logging is a minor objective nowadays. Forest products in the USA, consequently, are increasingly supplied by privately owned forests where inroads of certification have been very modest. Evidence of deleterious environmental impacts of logging may result in the cessation of logging of some tropical forests and perhaps even some boreal forests in the former USSR, but whole-sale abandonment of forest management for timber is unlikely. Therefore, it would be much more useful if researchers focused more on how to improve management rather than on trying to convince decision-makers that it should be stopped (Putz 2000; Putz and Zuidema 2008).

8.2.5 Focus

This study is also limited by its focus on the published literature. It disregards the knowledge and experience of forest managers and certification practitioners.

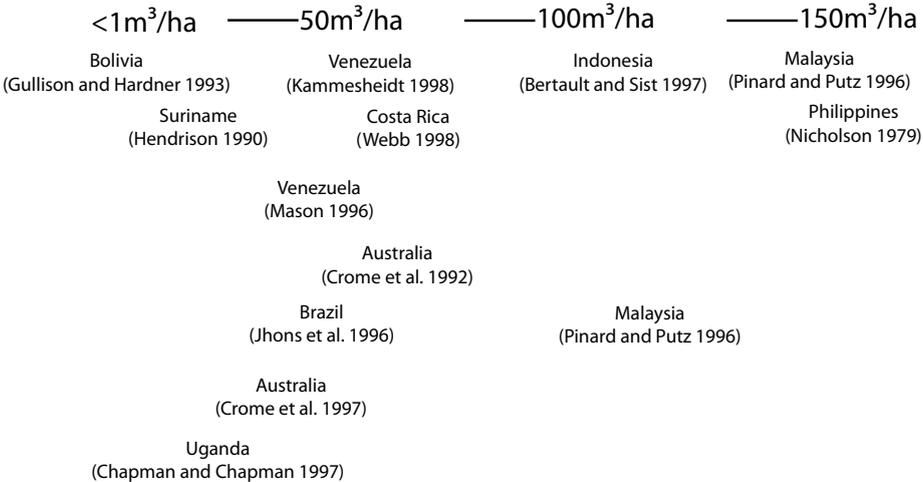


Figure 1. Logging intensities (m³/ha) for tropical forests. From Putz et al. 2001.

8.3 Assessing biodiversity impacts

Apart from the limitations to this study and the underlying literature discussed above, there are more conceptual challenges to the analysis of biodiversity and certification. These relate to the complex nature of both biodiversity and forest management, and to the different perceptions held by stakeholders about the value of biodiversity, their varying interests and the objectives of forest management.

Assessing the biodiversity impacts of forest management activities, certified or otherwise, is made challenging by the wide variety diversity of forests in which they are applied. Some managed forests are naturally low in diversity and are exceedingly resilient due to a history of major natural environmental disturbances (e.g., fire-maintained forests and typhoon forests). Globally, it is important to recognized that logging intensities span nearly three orders of magnitude, from 5-150 m³/ha (Figure 1). Then there is the challenge of differentiating between the primary impacts of logging and other forest interventions (e.g., erosion from logging roads and increased forest flammability) from the secondary impacts (e.g., use of logging roads by poachers and ignition of forest fires by land squatters). Whether forest management activities are restricted solely to logging or if pre- and post-harvest silvicultural treatments (e.g., soil scarification and liberation of future crop trees) are also applied can substantially affect biodiversity impacts. The final consideration in this litany of factors rendering generalizations difficult is that forest management practices can be applied with care (e.g., use of RIL practices) or in ways that are unnecessarily destructive.

8.3.1 Measurement problems

Several factors related to measurements done in the field, make it difficult to make generalizations about the effects of management practices on biodiversity.

Species specific responses

Different species show different responses to the management practices prescribed by certification systems. For instance, in response to the same management practice, even closely related species have been shown to decline in abundance, to remain stable, or to increase (Putz *et al.* 2000). Moreover, species responses are also dependent on the type of forest and the time of year; not always do species react in the same way when the management practice is implemented in a different forest or in a different season. This makes it difficult to generalize species responses to management practices. Most researchers therefore suggest that more research should be done with more species, more forest types, and more forest treatments applied at different times of year.

Long-term effects

Most of the studies we found in the literature were short-term, extending only a few years after logging at most. In some cases, modeling was used to project short-term data over longer time periods. Many researchers reported that more studies are necessary to evaluate the long-term effects (at least more than one logging cycle) of different management practices. For instance, in the many cases in which species

abundance reportedly decreased after disturbances, there is no way of knowing whether this response is a permanent phenomenon or whether the forest recovers its full complement of biodiversity over periods as short as 5-10 years. This focus on the immediate and short-term effects of logging and other forest management activities is understandable, given the difficulties in accessing areas much longer after treatment, and the lack of information about exactly what happened in the forest. Nevertheless, if we are to know whether implementation of the management practices required by certifiers actually serves to maintain high level of forest biodiversity, then these challenges will need to be surmounted.

One clear short-term effect of moderate levels of disturbance, at least in the tropics, is an increase in the number of species. This increase is associated with the increasing habitat diversity caused by the opening of the canopy, leading to a proliferation and influx of species of open habitats. This effect is confounding the interpretation of short-term data and may mask underlying negative responses of 'true' forest species (of conservation concern) to logging, which may only become apparent in the longer term.

Implementation of management practices

Codes of forest management practices vary by country, as do the details of how management practices are implemented. Even within a country the prescriptions might differ among forest types and geographical zones. Virtually all the studies we reviewed were conducted in a single forest and tested the effects of very site specific practices, which makes it difficult to make world-wide generalizations.

Are all species equal?

While many conservationists are concerned about maintaining rare, endangered, red-listed, or otherwise noteworthy species, forest managers need to be more concerned with maintaining the productive capacity of the forest. By taking species richness and abundance as a measure of biodiversity, each species is treated equally and the value of different species to the ecosystem or to society is not taken into account. Even though some studies focus on rare, threatened, or otherwise noteworthy species (e.g., the "charismatic megafauna"), in those studies in which numbers of species are tracked, all species are weighted equally. We also found scant attention to the roles that different species play in forest communities (e.g., seed dispersers and pollinators). Many studies reported that species composition changed after implementation of a management activity, for instance the replacement of forest interior species by species adapted to open conditions. If biodiversity is measured as species richness, the maintenance of biodiversity could go hand in hand with a loss in conservation value when rare, endemic species or species with crucial roles in the ecosystem disappear. This loss could go undetected when species richness is used as the parameter to describe biodiversity. An analogous loss of biodiversity in the sense of societal value occurs when species used by local communities as, e.g., food or medicine are lost and replaced by 'useless' species. Therefore, the choice of the biodiversity parameter strongly determines the outcomes of studies on the impact of management practices. These parameters must be established as a part of management planning

of each forest, and be included in monitoring protocols and certification standards for that management unit.

Certified forestry compared to what?

As outlined in the approach (section 2.4), our primary interest was to compare the biodiversity of forest subjected to selected good management practices with that of similar, 'normally' managed forests (as a proxy to non-certified forests). A secondary interest was to compare well-managed forests with similar, undisturbed forest. However, this does not cover the full suite of possible situations against which certified forestry could be compared. Other comparisons could be of relevance, but the literature is scant or not covered in this report.

In the tropics, for example, there are huge tracts of unlogged forests from which most game species have been extirpated by poachers (Bennett *et al.* 2002; Redford 1992). Another possible comparison is with areas of unmanaged forests subjected to episodic but extensive damage from wildfires associated with major droughts (Nepstad *et al.* 1999); focus in the literature has been on the synergy between logging and fires, but unlogged forests also burn. Finally, certified forests could be compared with those subjected to illegal logging, which is estimated to account for 50% of the timber harvested from tropical forests (Ravenel *et al.* 2004). While there is at least substantial anecdotal evidence that managers of certified forests control unsustainable hunting practices, fires, and illegal logging, there are few real data on which comparisons can be made.

All these comparisons would be useful to make judgments about the contribution and effectiveness of certified forest management to biodiversity conservation.

8.3.2 Forest managers, auditors or scientists? Practical challenges in estimating the impact of certified forest management

This report considers scientific studies about good management impacts. Conclusions about biodiversity responses certified forestry are, at best, extrapolated on the assumption that certification comes with the application of certain management practices, and that these generally lead to certain impacts on species. To judge whether conservation objectives are met in individual production forests is not a feasible option for researchers, except in individual cases. That means that the burden of proving the effect of forest management on biodiversity is on the shoulders of the forest managers and the auditors. If the task of scientists in demonstrating effects of forest management on biodiversity is daunting, then it is even more difficult for forest managers and auditors.

The fundamental challenges for forest auditors are lack of time and lack of information. The time limitation is general; given the direct costs of fielding auditing teams, it is unlikely that forest auditors will ever be allocated more than 2-3 days in managed forests to assess the direct and indirect impacts of a variety of forest management practices over areas that are often thousands of hectares.

The amount of information about management practices and impacts that forest managers are expected to provide certification auditors varies greatly among forest regions but also sometimes within the same region such as between large scale commercial operations and small scale community forest management areas. In many developed countries, for example, managers are expected to maintain permanent sample plots for monitoring growth and yield, as well as effects of management practices on forest composition. Many large forest corporations employ their own wildlife ecologists, botanists, remote sensing experts, and other environmental professionals that collect data that are useful to forest auditors. Simply the availability of high quality topographic maps and satellite images can greatly facilitate the auditing process.

The ability of forest managers to monitor biodiversity impacts obviously varies with the amount of scientific information available for their region and with the amount of this information to which they avail themselves. While managers can sometimes be faulted for not making adequate use of published information, in many places in the tropics, such information either does not exist or is not accessible to managers due to where it was published or the language of publication.

Generalizations about the biodiversity impacts of certified forestry are also difficult to make because the requirements of different certifiers vary. While efforts to align the standards of various major certification groups are making substantial progress, with increasing occurrences of joint-certification, there are still some fundamental differences that have potential biodiversity impacts (see also section 6.2).

8.3.3 Incorporating “acceptable change”

There is ample research demonstrating that all forest management activities have impacts on biodiversity but answering the question of “how much impact is acceptable?” is a societal not a scientific matter.

Some of the impacts of forest interventions are intentional, such as when trees are liberated from the growth-slowing and form-damaging cover by lianas. Some impacts are undesirable but unavoidable, such as the soil compaction that results from ground-skidding of huge logs by heavy equipment. But many of the impacts of forest management are both undesirable and avoidable. For example, given that many canopy-dwelling animals depend on lianas for food and inter-crown passage, only lianas on future crop trees might be cut, leaving the rest of the network intact (Mason and Putz 2001). Similarly, when RIL techniques are used, particularly the planning of skid trails and directional felling to facilitate log yarding, the amount of soil damage can be reduced substantially (Pinard et al. 2000).

Forest certification is an attempt at rationalizing societal concerns about the environmental (and social) damage done by loggers with societal demands for products from managed forests. The fact that the Forest Stewardship Council, the major certification system used in the tropics, is severely criticized by both environmental groups and forest industries reflect that the debate about acceptable change is fiercely fought and on-going.

The debate on acceptable change is strongly influenced by perceptions of the value of biodiversity, which calls for an approach that does not treat species equally as is done by standard scientific biodiversity metrics.

- As different species may be valued differently by different stakeholders, based on considerations of rarity, vulnerability, endemism, distinctness, economic usefulness, potential as pest, religion and many other considerations, the formulation of appropriate roles of production forests for conservation of biodiversity requires debate and negotiation at the local level (but not dismissing global interests).
- The results must be translated into practical management activities directed at specific, measurable biodiversity targets, subject to periodic revision to accommodate changes in value perceptions and in the state of biodiversity in the forest.
- What is needed from scientists to further inform the tradeoffs accepted by the FSC and other certifiers, is solid, quantitative, field-based evidence about the relation between forest management practices and species responses, and about further modifications of forest management practices required.
- Finally, biodiversity monitoring and audits of certified forest management should focus on these practical management activities and objectives rather than on general, unspecified biodiversity goals, which are almost impossible to measure and, if they can be measured, hard to interpret.





9

Conclusions and recommendations

1. Based on the published literature, there is no conclusive, quantitative evidence about the effect of forest certification on biodiversity.
2. **However, in general, good forest management practices associated with forest certification appear to benefit biodiversity in managed forests.**
3. There is a very high variation, both in forest management practices associated with certification and in responses between and even within species.
4. There is little quantitative evidence about the long-term effects of certified forest management on biodiversity.
5. There are few data on which to base the conclusion that certified forest management is sustainable in terms of biodiversity conservation at the level of populations and communities – we simply don't know.
6. Well-managed (certified) forests are not equivalent to undisturbed forests in terms of conserving (forest) biodiversity.
7. A systematic effort to study and understand the effects of certified forest management on biodiversity conservation – a major rationale for certification – appears to be lacking, both from the side of certification agencies and their clients, forest managers, and from the side of research community.
8. Research effort is skewed towards boreal and temperate forests, towards certain good forest management practices such as green tree retention and reduced-impact logging and towards vertebrates.
9. Our ability to assess biodiversity responses to certified forest management is limited by a lack of detailed species knowledge, variation in species responses, the absence or non-application of comprehensive research protocols suitable for establishing certification impacts, poor articulation of biodiversity objectives in forest management units, variation in certification standards between countries, forest types and certification systems, variation in auditing standards, among other things.
10. Some of these limitations are real and cannot be expected to be overcome. Yet, for reasons of transparency and credibility of forest management certification, it is important to demonstrate plausible relationships between certified management practices and forest biodiversity. The challenge for forest managers, certifiers and biodiversity researchers is to promote forest certification from a credible proposition to a demonstrated asset in the suite of instruments available for forest biodiversity conservation.
11. This requires:
 - a. A clear articulation of biodiversity objectives at the level of the forest management unit to be certified, reflecting the values attributed to biodiversity by different stakeholders, ranging from local to global.
 - b. A translation of these objectives into management activities with a demonstrated effect on the selected biodiversity objectives.

- c. Monitoring and adaptive management to cater for dynamics, change and uncertainty.
 - d. An independent and systematic research effort to understand and assess the relation between management practices and species responses that inform management decisions and trade-offs between incompatible management objectives. Appropriate reference against which to judge certified forest management practices should be incorporated in the research.
 - e. Auditing standards (and monitoring systems) that are designed to detect changes in the performance of specific biodiversity indicators and produce consistent results.
12. All species are not equal. Different stakeholders value different species in a different way according to their interests and values. Similarly, different species have different functions in the forest ecosystem according to the role they play and their abundance in natural forests.
 13. Biodiversity objectives should be consistent with the type of forest under management. An emphasis on 'high biodiversity' as a main desirable attribute of certified forest ecosystems disregards the importance of forests with a low diversity, forests with large-scale but infrequent disturbances, and other specific forest ecosystems.
 14. The evidence shows that there are also many species and ecosystems which are negatively affected by any form of logging. Therefore, sound conservation strategies must be grounded on an adequate mixture of protected areas and well managed forests.

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