

Chapter 3

Impacts of forest and land management on biodiversity and carbon

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CONTENTS

Abstract	54
3.1 Introduction	54
3.2 Impacts of management actions on biodiversity and carbon	56
3.2.1 Improving agricultural practice	56
3.2.1.1 Sustainable agricultural intensification	56
3.2.1.2 Agroforestry	56
3.2.1.3 Sustainable shifting cultivation	56
3.2.1.4 Fire management	57
3.2.2 Protection measures	59
3.2.3 Reducing impacts of extractive use	60
3.2.3.1 Reduced impact logging	60
3.2.3.2 Sustainable management of non-timber forest products extraction	62
3.2.3.3 Sustainable hunting in managed forests	63
3.2.4 Forest restoration, reforestation and afforestation	63
3.2.5 Landscape scale planning	65
3.3 Balancing opportunities and risks of different management actions for carbon and biodiversity	67
3.4 Monitoring to assess changes in forest carbon and biodiversity	69
3.4.1 Indicators of change in carbon	70
3.4.2 Indicators of change in biodiversity	70
3.4.3 Putting field carbon and biodiversity monitoring into practice	71
3.5 Conclusions	72
References	74

Abstract: Changes in the management of forest and non-forest land can contribute significantly to reducing emissions from deforestation and forest degradation. Such changes can include both forest management actions - such as improving the protection and restoration of existing forests, introducing ecologically responsible logging practices and regenerating forest on degraded land - and actions aimed at reducing drivers of forest loss and degradation through changes in agricultural practices. The impacts of changes to forest and land management on both carbon stocks and emissions, and biodiversity are often complex and non-linear. While REDD+ actions are always expected to contribute to reductions of carbon emissions or increases in carbon sinks, the outcomes for biodiversity can vary greatly, depending on the types of activities, the prior ecosystem state and the wider landscape context. Actions aimed at protecting existing forests from clearance and/or further degradation from fire and the overharvesting of timber and non-timber resources are likely to deliver both the greatest and most immediate benefits for the maintenance of carbon stocks and biodiversity. Where forests are already degraded or converted to non-forest uses, restoration and reforestation can generate rapid increases in carbon stocks but with varying impacts on biodiversity. There is the potential for negative biodiversity impacts if naturally regenerating forest or non-forest ecosystems are converted to plantation forestry. Resolving current uncertainties and information gaps requires developing effective programmes for assessing biodiversity impacts. These can be built on emerging monitoring, reporting and verification (MRV) systems for carbon in order to provide integrated guidance for the design and continuous improvement of national approaches to REDD+. In particular, information on both the spatial distribution of biodiversity and responses to different forms of management intervention can be used to guide strategic investments that achieve both significant emissions reductions and biodiversity conservation benefits.

3.1 Introduction

A wide range of *forest management*¹ approaches and actions can potentially contribute to the five broad activities (see Box 3.1) under the developing REDD+ mechanism, and many can support more than one of the REDD+ activities (Table 3.1). They include approaches directed at preventing human impacts on *forests*, reducing those impacts and accelerating the rate of recovery from anthropogenic disturbances.

Actions that address drivers of *deforestation* and *forest degradation* are essential to support the first two of the REDD+ activities. Thus, efforts aimed at improving agricultural practice may play an important role in REDD+ strategies. Sustainable agricultural intensification and improved management of existing production systems, including *agroforestry*, may help both to limit the increase in demand for new land (and consequent deforestation and forest degradation) and to reduce direct impacts such

as those from unsustainable *shifting cultivation*, the use of fire in land preparation and management, and the application of agrochemicals. Such improvements can also help to enhance *carbon stocks* of existing and new forests.

A critically important strategy for protecting forests from human impact is the establishment of formally protected areas or other conservation units. Such protection measures vary in their management approaches, which range from strict protection of *biodiversity* to allowing multiple uses, including limited extractive activities. They also vary in their *governance*, with some managed by government authorities and others managed privately or by local communities. Protected areas contribute to strategies to reduce *emissions* from deforestation or forest degradation or to conserve forest carbon stocks.

Reducing emissions from forest degradation will also depend strongly on actions to reduce the impacts of extraction of forest products including timber, fuelwood and other *non-timber forest products* (NTFPs). Such actions will include introduction of improved practices such as *reduced impact logging* (RIL), as well as promoting the sustainable use of non-timber forest products. Depending on national circumstances, such actions and the policies used to promote them can play an important role in REDD+ through their contribution to reducing forest degradation and promoting sustainable management of forests.

A broad set of management actions relates to the enhancement of forest carbon stocks through various forms of *forest restoration*, *reforestation* and *afforestation*. In addition to programmes of *assisted natural regeneration* in deforested or degraded areas, reforestation may also be achieved through establishment of tree *plantations*, using

Five REDD+ Activities

Box
3.1

REDD+

= Reducing Emissions from Deforestation and forest Degradation

+

Conservation of forest carbon stocks
Sustainable management of forests
Enhancement of forest carbon stocks

¹ All terms that are defined in the glossary (Appendix 2), appear for the first time in *italics* in a chapter.

Table
3.1

Relevance of management interventions to the five REDD+ Activities. Some interventions have a strong and direct role to play in a given REDD+ activity (**), while others may have less immediate relevance but may still play a role (*). For example, fire generally results in the first instance in forest degradation, but can in some cases eventually lead to deforestation; therefore, fire management potentially plays an immediate and important role in reducing emissions from forest degradation and plays a key role in ensuring the success of efforts to enhance carbon stocks and manage production forests sustainably. Fire management may also be relevant in reducing emissions from deforestation.

Forest Management Type & Management actions likely to be used in REDD	Relevance to REDD+ activities				
	Reduce emissions from deforestation	Reduce emissions from forest degradation	Carbon stock enhancement	Sustainable management of forests	Conservation of carbon stocks
Improving agricultural practice					
Sustainable Agricultural intensification	••	•	•		•
Agroforestry		•	•		
Sustainable shifting cultivation	•	••	•		•
Fire management	•	••	••	••	•
Protection measures	••	••	•	•	••
Reducing impacts of extractive use					
Reduced Impact Logging		••		••	
Efficiencies, alternative production, or substitution of fuelwood and NTFPs		••		••	
Hunting regulation		••		•	
Restoration/Reforestation					
Assisted natural regeneration	•	•	••	•	
Afforestation & reforestation primarily for wood/fibre production			••		
Reforestation primarily for biodiversity and ecosystem services		•	••		
Landscape scale planning & coordination	••	••	•	••	•

either *monocultures* or species mixtures, and either *native* or *introduced species*. In some cases *enrichment planting* may be used to modify the composition of existing forest and increase its value.

Finally, while any or all of these actions may potentially form part of REDD+ programmes and strategies, coordination and planning at landscape and broader scales are key to minimising negative impacts, and ensuring positive outcomes for both carbon and biodiversity.

This chapter aims to explore what is known about the impacts on carbon and biodiversity of these various management actions and approaches, and identify, insofar as possible, the circumstances under which they may have positive impacts on both, as well as the evidence for potential trade-offs between carbon and biodiversity objectives associated with their implementation. The analysis is based on the wide variety of indicators and measures that have been used in the published literature to summarise the impacts of human activities and management on forest carbon and biodiversity. Impacts on carbon are generally assessed using three primary indicators: forest area, carbon density and estimates of 'productive capacity', i.e., the quantity of carbon provided annually by the

managed landscape in the form of timber, fibre, energy and/or non-timber forest products. The types of indicators commonly used to assess biodiversity impacts are much more varied; here the focus is primarily on measures of *species richness* in particular taxonomic groups, measures of change in species composition relative to baseline conditions, including abundances of species of conservation concern, and measures of forest structure and landscape configuration that are understood to affect species-related trends. We also explore briefly the potential for unanticipated and indirect impacts, and the knowledge and research gaps that currently constrain informed decision making. The problems, shortcomings and complexities of indicator selection, and a brief outline of the methods available to assess carbon and biodiversity indicators and their changes with management activities are also discussed.

In addition to carbon and biodiversity outcomes, the assessment of REDD+ strategies requires consideration of the socio-economic issues (Chapter 4) and institutional and governance issues that may affect their implementation (Chapter 5).

3.2 Impacts of management actions on biodiversity and carbon

3.2.1 Improving agricultural practice

As discussed in Chapter 2, the growing human population and demand for agricultural products and the consequent expansion of both commercial and subsistence farming play a large role in causing forest loss (Kissinger et al., 2012). Four options for improving agricultural practice in order to limit the impacts of agriculture on forest carbon stocks are discussed: sustainable agricultural intensification, agroforestry, sustainable shifting cultivation and fire management.

3.2.1.1 Sustainable agricultural intensification

The problem of increasing agricultural production has traditionally been framed as a zero-sum game: increasing agricultural production will take away land that would otherwise be used for the conservation of carbon and biodiversity. Recently, however, it has been argued that, by increasing production per area, sustainable agricultural intensification (Royal Society of London, 2009) can reduce the need for additional agricultural land, reducing pressure for agricultural conversion of forests and therefore emissions from deforestation (Defries and Rosenzweig, 2010; Burney et al., 2010). However, intensification of agricultural production is typically achieved through increased use of agricultural inputs such as fertilisers and/or through mechanisation, both of which result in higher CO₂ emissions that must be considered in *carbon balance* assessments (Nabuurs et al., 2007).

There are also questions around the impact of agricultural intensification as a strategy for reducing pressure on biodiversity. On the one hand intensifying agricultural production to increase overall yield while avoiding further cropland expansion and deforestation or '*land sparing*' (Balmford et al., 2005; Green et al., 2005) is a promising approach to conservation in some circumstances (Phalan et al., 2011). On the other hand, the ecological impacts of intensive farming are often much greater than just the conversion of land it occupies (Matson and Vitousek, 2006). Intensive farming typically requires more irrigation, and fertiliser and pesticide inputs, which have downstream effects on *ecosystems* and cause pollution affecting aquatic and terrestrial biodiversity. Furthermore, intensifying agricultural production could lead to additional forest conversion by displacing people to other forested areas or by providing the economic incentives for migration into the area (Matson and Vitousek, 2006; also see Chapter 4, Section 4.4.1). There is also concern that biodiversity gains from intensification policies may be rapidly reversed if *habitat* that has been preserved is later made available for conversion (Ewers et al., 2009).

Therefore, while agricultural intensification may provide an appropriate strategy for reducing conversion pressures on forests in the short term, care is needed to assess its effectiveness in reducing emissions, taking account of the full carbon budget associated with the production

system and of the potential for indirect effects on land use (Nabuurs et al., 2007). Similarly, though forest area and *biomass* may be retained through this strategy, its biodiversity impacts - both at the local and at the landscape level - are much less well-understood and are dependent on the agricultural practices employed.

3.2.1.2 Agroforestry

Agroforestry may play a role in REDD+ both by reducing pressure on forests, through increasing agricultural productivity and providing some forest products, and by increasing tree cover in the agricultural landscape which also increases carbon stocks. It is estimated that globally, 558 million people utilise a wide variety of agroforestry systems (Zomer et al., 2009). Numerous inventories of smallholders' farms show that hundreds of indigenous and introduced tree species grow in tropical agroforests and can contribute to high levels of *species diversity* (Kindt et al., 2006; Bremer and Farley, 2010).



Intensively managed agroforestry systems combining timber, fruit and food crop production can increase agricultural productivity while easing forest conversion pressures. Photo © Manuel Bertomeu (courtesy of Springer)

The above-ground and below-ground carbon density of tropical agroforestry systems is estimated to be between 12 and 228 Mg ha⁻¹ with a median value of 95 Mg ha⁻¹ (Albrecht and Kandji, 2003). For smallholder tropical agroforestry systems, the potential *carbon sequestration* rates range from 1.5 to 3.5 Mg C ha⁻¹ yr⁻¹ (Montagnini and Nair, 2004). This estimate assumes the permanence of long rotation crops. For a given climatic zone, silvicultural practices, planting density, choice of species and their mixture, and length of rotation cycle are all known to influence carbon sequestration rates in agroforestry systems (Nair et al., 2010).

Agroforestry performs three key roles in biodiversity conservation (Schroth et al., 2004). First, agroforestry systems provide supplementary habitat for species that tolerate a certain level of disturbance; second, in certain cases, they reduce rates of conversion of natural habitat; third, they create a ‘matrix of connectivity’ between natural and/or modified forest remnants. In particular, complex agroforests (Michon et al., 2007), coffee agroforests (Komar, 2006), cocoa agroforests (Steffan-Dwenter et al., 2007) and silvopastoral systems (Murgueitio et al., 2011) may harbour high levels of both wild and agricultural biodiversity and offer much greater conservation value than agro-industrial monocultures or plantations of introduced trees. Overall, agroforests that mimic the structural and floristic diversity of native vegetation and rely less on pesticides and agrochemicals are likely to harbour more biodiversity and provide more associated ecosystem services than more intensively-managed agroforests (Gabriel and Tschardt, 2007; Letorneau and Bothwell, 2008). When agroforestry systems are located close to *primary forest* or modified forest habitat, key services such as insect pollination may fluctuate less and generate less variable crop yields as a direct function of pollinator diversity (Tylianakis et al., 2008; Klein, 2009). In contrast, a growing body of research (e.g. Phalan et al., 2011) argues that forest conversion to farming practices - even wildlife-friendly practices such as agroforestry - may have fewer conservation benefits than more productive agriculture that permits a greater preservation of *natural forest* within an agricultural landscape (also see Chapter 4, Section 4.4.1).

3.2.1.3 Sustainable shifting cultivation

Shifting cultivation (also known as ‘*slash-and-burn*’ or ‘*swidden*’ agriculture) is one of the most important land-use systems used by indigenous and local communities in most tropical forest regions (Coomes et al., 2000; Mertz, 2009; Ickowitz, 2006). It is embedded in the traditions and livelihoods of hundreds of millions of people worldwide (Parrotta and Trosper, 2012) and is often the only means available to ensure food security for the poorest people in rural landscapes (Padoch and Pinedo-Vasquez, 2010; Coomes et al., 2011). As discussed in Chapter 2, shifting cultivation makes a significant contribution to *greenhouse gas* (GHG) emissions from land-use change in the tropics. REDD+ can provide resources that make it possible for shifting cultivators to adjust their practices in line with *traditional knowledge* or other expertise to limit emissions and deliver biodiversity benefits, including through support of longer fallow periods, improvements in cropping and fallow management and, in some cases, introduction of fire-free approaches to site preparation.

Increasing the fallow phase, improving fallow management and/or reducing the time under production can improve the recovery of both biomass and biodiversity in sites cleared for shifting cultivation (see Chapter 2) and therefore could potentially achieve significant environmental benefits. Enrichment planting using fast-growing leguminous or other tree species to enhance soil fertility during the fallow phase (commonly used in traditional

shifting cultivation worldwide) could be more widely applied to accelerate forest regeneration and address declines in soil fertility (Ramakrishnan, 2007). However, the potential impacts of such interventions need to be evaluated in a wider landscape context as they may expand the area needed for cultivation and lead to new deforestation, compromising efforts to conserve primary and mature *secondary forest* areas. Evaluating the trade-offs and identifying an appropriate balance between these two approaches is a major challenge in developing REDD+ programmes. Another way of addressing shifting cultivation in REDD+ programmes in some regions would be to maintain the traditional rotational systems, but move towards fire-free management alternatives (Eastmond and Faust, 2006; Joslin et al., 2010) such as slash-and-mulch practices (Kato et al., 1999; Denich et al., 2004) and improved management of forest and crop residues (Sanchez et al., 1994; Ramakrishnan et al., 2012). In an integrated assessment of entire crop cycles in the Brazilian Amazon, GHG emissions in a traditional slash-and-burn plot were at least five times higher than in the fire free rotation system (slash-and-mulch) (Davidson et al., 2008). In some cases, fire-free management may require mechanisation and increased use of fertilisers, which have implications for GHG emissions, and also may limit the financial viability and the acceptability of these approaches for farming communities.

Shifting cultivation can also be replaced with more intensive farming (see above) or intensive commercial agriculture, as reported for Southeast Asia and elsewhere (Rerkasem et al., 2009; Ziegler et al., 2011). However, there are often strong arguments for maintaining shifting cultivation systems, given their importance for local livelihoods, food security, cultural identity, and environmental benefits compared to more intensive cultivation systems (Rerkasem et al., 2009; Padoch and Pinedo-Vasquez, 2010; Dalle et al., 2011; Barlow et al., 2012; Parrotta and Trosper, 2012).

3.2.1.4 Fire management

Whether it originates from shifting cultivation or from other land management practices, anthropogenic fire is an increasingly important cause of forest degradation and associated carbon emissions in many tropical forests (see Chapter 2). In tropical rainforests, fire intensity, the resulting tree mortality, and direct and indirect emissions are much higher when fires enter forests that are already heavily degraded (Holdsworth and Uhl, 1997; Cochrane, 2003). In addition to lowering landscape-level carbon stocks and contributing to increased emissions to the atmosphere, the reported increase in forest fires in the humid tropics severely threatens the long-term permanence of carbon stocks in undisturbed primary forests, logged forests, and forest regeneration and reforestation projects (Barlow et al., 2012).

Reducing forest fires becomes increasingly important for carbon management and REDD+ objectives if burned forests are unable to recover their original biomass within the average fire-return interval. Fires can also affect

REDD+ activities that promote the regeneration of forests following deforestation and degradation. Stickler et al. (2009) argue that the conversion of agricultural lands to allow forest regeneration will generally reduce the occurrence of fire, but this may be optimistic as regenerating forests are especially flammable (Ray et al., 2005) and could even increase the chance of transmitting fires to other land-uses.



Severe recurrent fire events can lead to functional deforestation, as seen in this picture of forests burned in the dry season of 2007 in the municipality of Querencia, located in the state of Mato Grosso in the southern Brazilian Amazon. Photo © Jos Barlow

The most obvious strategy for reducing large scale fires in tropical forests is to reduce deforestation. This has the potential to: i) reduce *fragmentation* rates, yielding fewer new forest edges that dry faster than forest interiors, are adjacent to managed areas, and are more prone to burning (Alencar et al., 2006; Cochrane and Laurance, 2002); ii) reduce agricultural fire use, if intensive agriculture is favoured over extensive agriculture (Angelsen, 2010) and iii) help prevent reductions in regional rainfall (Andreae et al., 2004; Eltahir and Bras, 1994).

Most tropical forest fires are caused by agricultural fires escaping into surrounding vegetation (Uhl and Buschbacher 1985), therefore fire can continue to remain a problem even in areas with little deforestation (Aragão and Shimabukuro 2010). Reducing the prevalence of agricultural fires requires both effective monitoring using remote sensing (Souza et al., 2005; Alencar et al., 2011) and improvements in their management, or their substitution by fire-free agriculture (Denich et al., 2005). Improved management of agricultural fires can be achieved through training, enforcing legislation, or new incentives. For example, Peskett et al. (2011) describe a case where

REDD+ payments provide incentives for local people in Uganda to maintain 100m forest buffers around a protected area, helping protect it from fire.

In the longer-term, support from REDD+ programmes could provide the capital and technical investments necessary to facilitate the shift toward fire-free agricultural practices (Palmer, 2011), such as mechanised land preparation, slash-and-mulch, perennial agriculture (with less frequent land preparation requirements), or intensive pasture management (Eastmond and Faust, 2006; Tschakert et al., 2007). However, this will be difficult to implement in active frontier zones where fire-dependent slash-and-burn agriculture is most common, where technology and non-fire alternatives such as mechanisation are hard to deliver, and where landowners lack the secure land tenure required for effective payments (Hirsch et al., 2010; also see Chapter 4, Section 4.2.3).

In addition to its obvious value for reducing GHG emissions, limiting the incidence of fire can potentially yield substantial biodiversity benefits, especially in humid tropical forests, by reducing the well-documented adverse impacts of fire on biodiversity, including those on tree species composition (Barlow and Peres, 2008; Cochrane and Schulze, 1999; Slik et al. 2002; 2010), birds (Barlow and Peres, 2004a; Barlow and Peres, 2004b; Adeney et al., 2006), butterflies (Cleary, 2003) and large vertebrates (Lees and Peres, 2006; Michalski and Peres, 2005). Recurrent forest fires cause the loss of most biodiversity of high conservation concern, removing 72 percent of the bird species recorded in the understorey of unburned Amazonian forests (Barlow and Peres, 2004a; Barlow and Peres, 2004b) and causing significant changes in species composition of forest birds in Sumatra (Adeney et al., 2006) and butterflies in Borneo (Cleary, 2003). Forest species recovery is slow after fire, and lag-effects in biodiversity responses can cause the species composition of burned forests to diverge from that found in unburned forests over time (e.g. Adeney et al., 2006; Barlow and Peres, 2004a) because species respond to changes in the structure and composition of the regenerating understorey vegetation. Consequently, the immediate biodiversity benefits to be gained by protecting heavily degraded humid forests from future fires may be limited, and management priorities should be focused on preventing the encroachment of fires in relatively undisturbed areas of forest. However, longer-term benefits may exist if degraded forests (which are the only type of forest left in many areas) recover carbon and some biodiversity over decadal time-scales, if restoration schemes can speed recovery, or if these forests help provide connectivity to forest biodiversity at a landscape level.

While the focus here is on efforts to reduce fires in humid forest ecosystems, fire management can also play a very important role in retaining carbon stocks and native biodiversity in tropical dry forests, through low intensity prescribed burning (Ryan and Williams, 2011). However, it is important to note that suppressing fires in tropical dry forests and savannah ecosystems that have natural fire regimes can lead to catastrophic ecological damage following the build-up of large fuel loads (Bond and Parr,

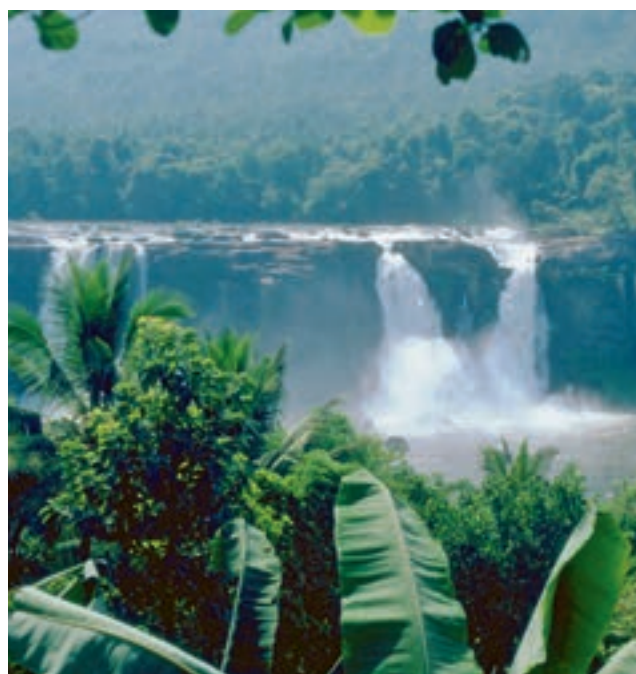
2010). Woody biomass in fire-prone ecosystems may nevertheless be preserved or enhanced by prescribed, low-intensity burning.

3.2.2 Protection measures

Protected areas are one important strategy for conserving forests and their biodiversity. Their spatial coverage has been increasing globally over the past twenty years (Jenkins and Joppa, 2009; Butchart et al., 2010), but they can vary considerably in the ways they are established, governed and managed (Dudley, 2008). They may be established and recognised in formal legal terms in relation to international, national or local laws, or through customary law, covenants or private trusts and policies, and may be governed by authorities ranging from state agencies to local communities, civil society organisations or private land holders. Regardless of the type of designation or governance, protected areas vary in their management objectives from strict protection, strongly limiting human activity, to management specifically in relation to the interactions between people and nature, including sustainable use of natural resources.

In an effort to standardise the global use of the term protected area, the International Union for Conservation of Nature (IUCN) developed a system of six different protected area management categories where those with more stringent levels of protection fall into categories I–IV, while those aimed at sustainable resource use are within categories V and VI (Dudley, 2008). However, the extent to which countries apply the IUCN categories system varies widely (e.g., Burgess et al., 2007) and a range of alternative forms of governance exists, including co-management approaches and community management of protected areas. The wide range of different protected area types and terminologies complicate global comparison of protected area extent and effectiveness (Schmitt et al., 2009; Chape et al., 2005).

Across the (sub-) tropical moist broadleaf forests, protected areas of all types (with and without IUCN categories) are estimated to cover an average of 28 to 29 percent of forest area, yet there is more than twice as much forest



Athirapally waterfalls on the Chalakudy river, Vazhachal forest division, Kerala, India. Photo © John A. Parrotta

protected area coverage in the Neotropics (38 percent) as in Africa or Asia. Protected areas with sustainable use (IUCN categories V and VI) are more prominent in the Neotropics than in the other regions (Nelson and Chomitz, 2011; UNEP-WCMC et al., 2008), and forest protection through Indigenous Reserves occurs only in the Neotropics (about 85 million ha; Nelson and Chomitz, 2011). In the other (sub-) tropical forest *biomes*, the percentage of forest protection is much lower (Table 3.2), though there are large variations between regions (Schmitt et al., 2009). The lower protected area coverage of the drier (sub-) tropical forest types is mostly due to the fact that these forest areas are often less remote than rainforest areas and highly suitable for agriculture or cattle grazing, which makes establishing protected areas more contentious (Miles et al., 2006; Silva et al., 2006; Boucher et al., 2011).

Protected area gap analyses (Dudley and Parish, 2006) have been used to assess the extent of protected area coverage for threatened species (Brooks et al., 2004; Rodrigues et al., 2004), conservation priority areas (Soutullo et al., 2008) and forest types (Schmitt et al., 2009). Since the establishment of protected areas has often been guided by lobby groups, politics and opportunity (Chape et al., 2005; Halpern et al., 2006), protected area gap analysis is a strong tool for making conservation planning more science-based and systematic (Schmitt, 2011; Meir et al., 2004; Margules and Pressey, 2000). There is, however, much controversy about how much protected area coverage is enough for different species and ecosystems (Carwardine et al., 2009; Tear et al., 2005; Svanicara et al., 2005; Rondinini and Chiozza, 2010).

Scharlemann et al. (2010) estimated that in 2000, almost 20 percent of the humid tropical forest biome was

Percentage of forest area (at 10 percent tree cover threshold) in tropical and sub-tropical forest biomes (see Olson et al., 2001) within all types of protected areas (UNEP-WCMC et al., 2008).

Table 3.2

Biome	Protected Area Coverage of Forests
(Sub-)Tropical Moist Broadleaf Forest	29%
(Sub-)Tropical Dry Broadleaf Forest	16%
(Sub-)Tropical Coniferous Forests	8%
(Sub-)Tropical Grasslands, Savannas and Shrublands	15%

under strict protection (defined in this analysis as only IUCN Categories I and II) and contained about 3.5 per cent of global terrestrial carbon stocks. However, loss of forest cover from these protected areas between 2000 and 2005 may have resulted in emissions as large as 1 Pg CO₂. In their study this was equivalent to about half of the emissions coming from tropical forests outside protected areas during the same time interval.

The global distribution of protected areas says little about how effective they are in achieving their management objectives, and there is uncertainty regarding the number and degree to which some of the protected areas in the tropics might be ‘paper parks’, i.e. protected only on paper but not in practice (e.g., Bonham et al., 2008; Bruner et al., 2001). Generally, the assessment of biodiversity impacts of protected areas is a complicated task, especially in (sub-) tropical ecosystems with high species richness. The same difficulties and challenges that apply to developing and selecting biodiversity indicators and monitoring schemes for REDD+ (see Section 3.4), also apply to protected areas. Hence, data on the biodiversity impacts of protected areas are often not available or not comparable regionally and globally due to lack of standardised monitoring tools for impact assessment (Chape et al., 2005). However, systematic assessment of protected area management effectiveness covering not only biodiversity but also planning and governance issues is increasingly being applied (Hockings, 2003; Hockings et al., 2006; Stoll-Kleemann, 2010; Leverington et al., 2008; Porter-Bolland et al., 2012).

Forest protected areas can offer potential synergies for biodiversity and carbon objectives (Campbell et al., 2008). For instance, it is well established that all types of forest protected areas reduce deforestation relative to non-protected surroundings (Naughton-Treves et al., 2005) but they do not all eliminate deforestation (DeFries et al., 2005; Nagendra, 2007; Barber et al., 2012). Protected forest areas are often established in remote locations where drivers of forest conversion are absent or limited and statistical ‘matching’ methods show that they may not have as large an effect in reducing deforestation as previously thought (Andam et al., 2008; Joppa and Pfaff, 2009). The degree of protection is also relevant, with strictly protected areas playing an important role in conservation of both carbon stocks (Scharlemann et al., 2010) and biodiversity. On the other hand, protected areas that allow for the sustainable use of forest resources are expected to have positive biodiversity impacts, but their carbon impacts may be more variable depending on the kind of management practices applied. Where resource use includes timber harvest, landscape-level carbon stocks will be reduced relative to the carbon density in the primary forest (see below).

Strict forest protected areas and those with multiple uses are recognised as two widely separated points on a continuum of conservation strategies involving a mix of top-down enforcement of regulations and local co-management. While protected areas are likely to remain a cornerstone of biodiversity conservation, landscape-scale land-use planning that incorporates diversely managed

and governed protected areas is also deemed essential (DeFries and Rosenzweig, 2010), especially in areas with high population pressure.

3.2.3 Reducing impacts of extractive use

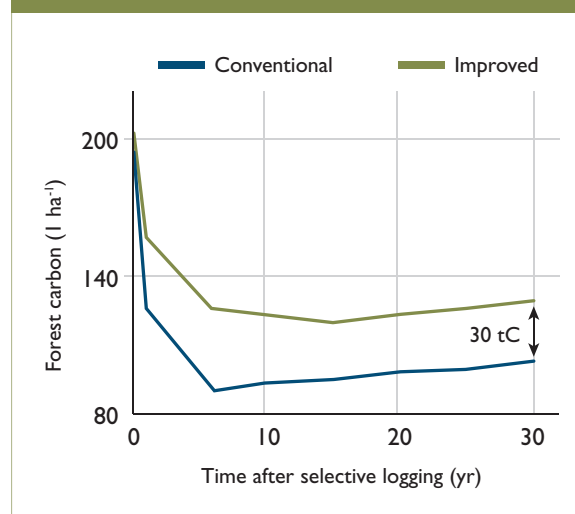
The International Tropical Timber Organization (ITTO; Blaser et al., 2011) estimates the size of the natural tropical permanent forest estate (PFE) to be about 760 M ha, comprising 400 M ha of production forest and 360 M ha of protection forest (designated to remain as forest for purposes other than production, such as soil conservation and watershed protection). These forests supply a wide range of important products for both commercial and subsistence use, and the extraction of these products generally leads to greater or lesser degrees of forest disturbance and associated carbon emissions. REDD+ strategies will need to include approaches for reducing these impacts and managing forests sustainably, including through the application of appropriate management, policy and tenure frameworks that can support sustainable timber harvesting and other forms of sustainable management.

3.2.3.1 Reduced impact logging

Vast areas of tropical forests lying outside of protected areas are either being logged or are likely to be logged in the near future, resulting in many cases in forest degradation and associated carbon emissions (see Chapter 2). Therefore, forest management for timber is a likely focus for REDD+ action. The impact of timber harvesting varies significantly with the practices employed, including logging intensities, felling practices and harvesting strategies, rotation cycles, length of seasonal closure periods and post-harvest interventions (Fimbel et al., 2001; Putz et al., 2008a). Also, logging activities are often followed

Substantial reductions in carbon loss from improved forest management at scales of a hectare of Malaysian forest. (Putz et al., 2008b)

Figure 3.1



by an increase in human activities that can propagate fire (such as shifting cultivation, charcoal production and fires used in hunting or harvesting other forest products). Conventional or poorly-managed logging practices commonly result in the loss or damage of some 10-20 trees for every tree that is felled in tropical forests. Applying reduced-impact logging (RIL) techniques, which include reducing harvest intensity, carefully managing access and removal routes and using well-planned directional felling, can reduce this collateral damage by at least 50 percent (Uhl and Vieira, 1989; Putz et al., 2008b).

Employing RIL techniques can reduce carbon emissions from logging by 30 - 50 percent compared with conventional timber harvesting (Pinard and Putz, 1996; Bryan et al., 2010; Medjibe et al., 2011; Miller et al., 2011; Figure 3.1). If adopted globally this would be approximately equivalent to an avoidance of 10 percent of total emissions from deforestation (Putz et al., 2008b). Nevertheless, any form of extractive timber use reduces landscape-level carbon density (Putz et al., 2012), with greater reductions in landscapes with higher logging frequency and/or higher logging intensity. Sustaining carbon stocks requires a combination of lower logging intensities and longer cutting cycles than is currently employed anywhere (Blanc et al., 2009; Mazzei et al., 2010). For example, Sasaki et al. (2012) found that landscape-level *above-ground biomass* carbon stocks were maintained in reduced impact logging systems with 50-year return intervals. Shorter return intervals, or the use of conventional logging methods, both resulted in substantial reductions in carbon stocks, and these losses were ongoing even after 50 years of simulated management scenarios.

Even under RIL the long-term recovery of both timber and carbon stocks may not be possible without a combination of reduced logging intensity (i.e. trees extracted per hectare) and active management interventions to aid regeneration, including enrichment planting and the prevention of fire. For a major RIL enterprise in the Amazon, Mazzei et al. (2010) estimated that while above-ground biomass recovery was possible within 15 years at a maximum logging density of three trees ha^{-1} , higher logging densities (6-9 trees ha^{-1} ; comparable or lower than RIL practices in the region and elsewhere) would require a much longer recovery period than the 30-year cutting cycle required by law in Brazil and in other countries.

Any form of periodic logging will reduce carbon density. The amount of carbon removed in each intervention, the amount of collateral damage to remaining trees, the frequency of intervention, and the rate of regrowth affect the landscape-level carbon density and biodiversity (Kurz et al., 1998; Sasaki et al., 2012). Net emissions to the atmosphere occur during the transition from primary to managed forest, but once landscape-level carbon stocks have reached the lower, sustained carbon levels then net biotic emissions are near zero, while the landscape provides an annual supply of timber. A proportion of the carbon removed through harvest will accumulate in wood products or landfills. However, this transition can take decades, and sustainable carbon levels are only achieved when disturbance pressure is not increasing.

In general, selectively-logged forests provide habitat for significantly more forest species than either *planted forests* or forests that are regenerating on cleared land (Gibson et al., 2011). Indeed, when comparing only patterns of species richness, logged forests can be indistinguishable from *primary forest* (Putz et al., 2012) and are capable of retaining substantial biodiversity even after severe and repeated logging. For example, Edwards et al. (2011) found that over 75 percent of species of birds and dung beetles found in unlogged forest persisted in twice-logged forest.

Differences in how forests are managed determine the impacts of logging on wildlife through changes to the structure and composition of the forest, fragmentation of the canopy, soil compaction and alteration of aquatic environments (ITTO and IUCN, 2009). In general broad patterns of wildlife response can be explained by differences in the intensity of logging activity (as well as the recovery time in between studies; Putz et al., 2001). Within any one group it is invariably the forest-dependent and specialist species as well as *endemics* that decline, while generalist and omnivorous species are unaffected or even increase in abundance and diversity. There is an urgent need for more long-term studies to better understand medium and longer-term response trajectories.

In general terms, biodiversity responses to logging are consistent across different forest types, with a high retention of species richness, often accompanied by marked shifts in species composition in conventionally logged sites, and more subtle yet still measurable impacts on particularly sensitive species in RIL areas. For example, RIL and low intensity logging in Borneo had a less marked effect on dung beetle communities in Borneo than conventional logging, although there were still noticeable shifts in species composition (Davis, 2000; Slade et al., 2011).

Even under reduced-impact harvesting regimes, over time managed forests may diverge from undisturbed forests in species composition and community structure (Keller et al., 2007; Putz et al., 2012). Depending on their severity, such changes can increase *vulnerability* of forests to both exogenous (e.g. fire) and endogenous (e.g. shifts in species composition) threats. Although they cannot completely mitigate negative impacts, RIL techniques can dramatically improve the retention of carbon and biodiversity as well as help maintain forest *resilience* to future impacts (e.g. from fire). Reduced impact logging techniques need to be applied as a package of measures that include pre-, during and post-logging interventions at stand and landscape scales. An upper limit in (sub-)tropical forests of approximately five harvested trees ha^{-1} , may be necessary to avoid significant changes in forest structure and diversity at the stand scale, as well as delayed biomass recovery (Aguilar-Amuchastegui and Henebry, 2007; Mazzei et al., 2010). At both stand and landscape scales retention of sufficiently large, undisturbed areas of forest, is necessary to provide refugia for those species that are particularly sensitive to logging activities. Conserving logged and degraded forests, especially in regions that have suffered large-scale deforestation, can represent an important investment for forest conservation as they can still retain appreciable carbon and biodiversity values

(Fisher et al., 2011), and can be managed to enhance carbon stocks (e.g. through enrichment planting, liberation thinning of vines and bamboos) with little detrimental effect on biodiversity (Ansell et al., 2011) and at much lower costs than replanting programmes (Sasaki et al., 2011).

3.2.3.2 Sustainable management of non-timber forest products extraction

Non-timber forest products (NTFPs) include resins, exudates, bark, foliage, fruits, seeds and fuelwood collected from forests (or sometimes cultivated within or outside of forests) for both consumptive and non-consumptive uses. They range from products traded internationally, through those traded in local or regional markets, to resources harvested for subsistence consumption or as supplementary resources during periods of scarcity.

Sustainable management of NTFPs plays two potential roles in REDD+. Firstly, it can help reduce NTFP extraction that contributes to forest degradation and associated emissions. Secondly, sustainable exploitation of NTFPs can contribute to reducing degradation and deforestation caused by other factors, by (a) increasing the value of standing forests and thereby reducing pressures on them, and (b) providing alternative sources of income to activities that deplete forest carbon stocks.

growth rates and large populations are more able to withstand repeated harvest than those with opposite attributes. Non-timber forest product species that depend on generalist species or abiotic mechanisms for pollination and/or dispersal are also more resilient to repeated harvest. In contrast, NTFPs involving harvesting of whole individuals, or those derived from species with restricted habitats, low adult population densities or growth rates, or specialist biotic relationships, generally have low potential for sustainable harvest. Non-timber forest products harvested in forests with high levels of solar radiation in the understorey recover faster to pre-harvest conditions than those from darker, primary forest understoreys (Ticktin and Nantel, 2004).

Interactions with other management systems may further influence the sustainable harvest potential of NTFPs (Guariguata et al., 2010). In Mexico a decreased frequency of fallow periods in slash-and-burn fields affected the supply of the Sabal palm (*Sabal palmetto*; Pulido and Caballero, 2006). Silvicultural treatments that reduce canopy cover can promote regeneration and/or growth of light demanding NTFP species (Salick et al., 1995; Wadsworth and Zweede, 2006; Peña-Claros et al., 2008). Good practice RIL for timber harvesting may facilitate NTFP management objectives. For example, liana cutting to minimise logging damage to residual trees (Putz et al., 2008a) could enhance fruit production in NTFP-bearing trees (Kainer et al., 2006). However, silvicultural practices can also have negative impacts, as in Indonesia, where a requirement to slash all undergrowth and climbers in logging compartments annually for five years after logging to promote timber species regeneration has adverse impacts on locally-important NTFPs (rattans, food and medicinal plants; Sheil et al., 2006; Meijaard et al., 2005). When the same tree species provides both timber and NTFP values (e.g., in Pará, Brazil, 47 percent of the timber species currently traded also have non-timber uses; Herrero-Jáuregui et al., 2009) conflicts can arise between users (Schulze, 2008; Tieguhong and Ndoye, 2007).

Some NTFPs, such as the Brazil nut tree, whose productivity is largely contingent on the presence of forest cover (Ortiz, 2002), hold promising opportunities for developing REDD+ schemes while promoting local livelihoods (Nunes et al., 2012; also see Chapter 4, Section 4.4.3). Options for enhancing sustainability of other NTFP use include regulation or modification of both timber and NTFP harvest. Where the economic and social values of NTFPs exceed the timber value of a site, regulation of logging may protect the NTFP and help maintain forest carbon stocks. Spatial separation of management units for timber and NTFPs can also help, as in the case of crabwood (*Carapa guianensis*), where seasonally-flooded forest is designated for seed collection while timber extraction is permitted in *terra firme* forests (Klimas et al., 2012). The application of RIL methods may also help in sustaining yields of NTFPs, as suggested for the Brazil nut tree, which coexists with valuable timber species across the Western Amazon (Guariguata et al., 2009). Restricting NTFP harvesting to specific size classes can also enhance sustainability (Ticktin, 2004). Finally, enrichment



Non-timber forest products, Badaling, China.
Photo © Alexander Buck

The potential for increasing the sustainability of NTFP use depends on the products extracted and the characteristics of the species and the forest. Harvesting of fruits, seeds and *dead wood* has the highest potential for sustainability. Long-lived species (e.g. Brazil nut, *Bertholletia excelsa*; Zuidema and Boot, 2002) and those with fast

planting of NTFP species (e.g., the understory bromeliad *Aechmea magdalenae* or Açai palm, *Euterpe oleracea*) within forests may both enhance sustainability of harvest and provide incentives to minimise forest conversion (Marshall et al., 2006; Pinedo-Vasquez et al., 2012), thus contributing to achieving REDD+ objectives.

3.2.3.3 Sustainable hunting in managed forests



Black-headed squirrel monkey (*Samiri vanzolinii*), endemic to the Mamirauá State Sustainable Development Reserve. Photo © PJ Stephenson

Animals perform a large number of key *ecosystem functions* and *processes*, including nutrient recycling and seed dispersal, which are key to maintaining the ecological integrity, species composition and productivity of forest ecosystems (see Chapter 2). The biodiversity benefits provided by forests protected under REDD+ will be reduced if they are allowed to become ‘empty forests’ due to unsustainable hunting activities (Redford, 1992; Collins et al., 2011). While the immediate impacts on biodiversity and carbon of strategies aimed at reducing hunting pressures are much smaller than other REDD+ activities, over long timeframes controls on over-hunting of game animals can affect biodiversity and carbon stocks.

Improvements in the sustainability of hunting practices can be made through a variety of interventions including changes in legislation, voluntary agreements by local communities and landowners and changes in individual behaviour (Millner-Gulland and Rowcliff, 2007). Over-exploitation of game animals in forests often accompanies the expansion of settlements to previously uninhabited areas of forest, including through road building and timber harvesting operations. Changes in the management of timber harvesting operations can mitigate the

negative impacts of their activities on wildlife by controlling and managing bushmeat hunting, including through the provision of affordable protein alternatives for forest workers and their families, preventing the use of company vehicles for bushmeat hunting, limiting access to forest roads to company vehicles and rendering roads that are no longer required for logging impassable for vehicles (Nasi et al., 2012). Through local enforcement measures, companies can ensure that their workers hunt legally (with proper licences and permits) and impose penalties or fire those who break the law. Forest management enterprises may also formalise hunting zones within their management plans and offer priority access to the original inhabitants of the area (Poulsen et al., 2009; ITTO and IUCN, 2009). Other suggested practices include banning commercial hunting in timber concessions, establishing conservation zones within the concession where hunting is forbidden, prohibiting unselective hunting methods such as snare hunting and trap hunting, and producing educational and information materials for both the public and staff (Meijaard et al., 2005).

3.2.4 Forest restoration, reforestation and afforestation

Increased carbon stocks and enhanced biodiversity can also be achieved through activities aimed at reversing forest loss and degradation, such as restoration, reforestation and afforestation. Globally as much as two billion ha of land are estimated to be available for forest restoration (Minnemeyer et al., 2011). In 2010 an estimated 264 million hectares of planted forests existed worldwide (about 7 percent of the total forest area) of which around 76 percent had timber and fibre production as their primary function (FAO, 2010).

‘Passive’ restoration approaches allow secondary forest development to proceed with minimal human input through natural regeneration (*assisted natural regeneration*) and by suppressing the causes of ongoing forest degradation. Under such approaches, forest biomass and tree species richness may begin to resemble those of mature forests after 30 to 40 years of secondary *succession* depending on the intensity and severity of past land use and the distance to patches of undisturbed forest that act as sources for plant and animal colonists (Guariguata and Ostertag, 2001; Lamb et al., 2005). The area of tropical secondary forests has increased notably over the last few decades, with much of this increase attributed to natural regeneration following forest clearance (Corlett, 1994; Chazdon, 2008), for example on abandoned agricultural lands. Secondary forest development and their influences on carbon and biodiversity are discussed in Chapter 2 of this report (see Section 2.5.5).

‘Active’ approaches include tree planting or seeding (i.e., *planted forests*) to expand forest cover on non-forest lands (*afforestation*), or to re-establish forest cover on deforested or degraded forest lands (*reforestation* and *forest restoration*). Depending on the management objectives and site conditions prior to planting, and the planted species, all of these approaches can yield biodiversity

benefits and enhance the provision of a range of ecosystem goods and services. Management practices associated with planting or seeding typically include site protection, more or less intensive site preparation, weed and fire control, and fertilisation (ITTO, 2002). Afforestation and reforestation typically involve planting of one or a limited number of tree species. Production of pulp and timber has been the primary purpose of plantations using fast-growing - usually introduced - species, most commonly *Acacia*, *Eucalyptus* and *Pinus* species (Lamb et al., 2005). Plantations that include a mixture of native and introduced species are used for watershed protection and erosion control (CIFOR, 2002). Enrichment planting using native species in degraded forests can help to stimulate natural succession (Whisenant, 2005).

Planted forests established for restoration purposes (i.e., to regain original forest structure, ecological functions and species composition, or to enhance landscape connectivity) usually involve the use of larger numbers of native tree species and reliance on forest successional processes. While favourable carbon and biodiversity outcomes may be achieved, the timing and magnitude of these results are uncertain, and complete restoration of pre-disturbance ecosystem conditions is unlikely. The importance of securing resilient, resistant and dynamic ecosystems, under changing land-use conditions, can at times justify not attempting to recreate reference ecosystems (Lugo 1997; Ewel and Putz, 2004; Harris et al., 2006).

The choice of restoration or plantation techniques and their effects on carbon and biodiversity (at both the site and landscape level) will be dictated to a large extent by the ultimate objectives (i.e., the expected or desired *ecosystem services*) for the planted or restored forest (Sayer, 2005) as well as the local ecology.

Several other considerations play a role in determining both carbon and biodiversity impacts regardless of the techniques applied to increase tree cover, namely:

- Prior land use and degree of degradation and forest fragmentation both at the site and landscape levels (e.g. Carnus et al., 2006; Chazdon, 2008; Omeja et al., 2012);
- Choice of species (or species mix) (e.g. Brockerhoff et al., 2008; Montagnini and Nair, 2004; Kanowski and Catterall, 2010);
- Location and spatial scale of forest planting or restoration activities within the landscape (Brockerhoff et al., 2008; Lamb et al., 2005);
- Silvicultural methods used for site preparation, planting and subsequent management activities (e.g. Montagnini, 2005); and,
- The time scale over which the restoration or planting effort is sustained, monitored and managed (e.g. Mansourian et al., 2005);

Landscapes comprised of planted forests typically have lower carbon stocks than those of primary or mature secondary forests, but their carbon stocks are higher than in non-forest lands or in highly degraded forests. When accounting for carbon in planted forests the carbon in

biomass and in litter, dead wood and *soil carbon* pools needs to be considered, as gains in one carbon pool may be offset by losses in the others (Russell et al., 2010). For example, afforestation is frequently assumed to enhance carbon stocks, but in some non-forest ecosystems, such as grasslands, savannahs and shrublands, gains in biomass carbon stocks may be offset, at least in the short term, by soil carbon losses through increased soil respiration and soil loss (Guo and Gifford, 2002; Resh et al., 2002). While afforestation of these non-forest ecosystems may in some circumstances yield net carbon benefits in the long term, they can have lasting adverse impacts on biodiversity. Afforestation of severely degraded land can however, provide net benefits both for carbon and biodiversity.

For a given site, the choice of species planted will determine changes in biomass and soil carbon pools, principally due to growth rate differences among species, affected by site conditions (Loaiza et al., 2010; Kanowski and Catterall, 2010; Silver et al., 2004) and by biodiversity effects on ecosystem function (see Section 2.2). The silvicultural methods used for site preparation, planting and subsequent plantation management will also significantly influence carbon stocks. Site preparation generally reduces soil carbon pools during the first five years (Hartley, 2002; Lindenmayer and Hobbes, 2004; Carnus et al., 2006). Depending on the condition, prior land use, size of soil carbon stores of the area planted, and management practices, soil carbon can decline in the first 10 years of growth (Russell et al., 2004; Resh et al., 2002) and then increase in subsequent years (Zheng et al., 2008; Paul et al., 2002).

Carbon sequestration rates from restoration plantings may be expected to be similar to, or somewhat greater than, that in secondary forests of similar ages on abandoned agricultural lands and pastures, which have been estimated to reach up to 3.6 Mg C ha⁻¹ yr⁻¹ in above-ground biomass, and 1.3 Mg C ha⁻¹ yr⁻¹ in soils during the first 20 years (Silver et al., 2000). A comparative study of 13-14-year old monoculture plantations of native rainforest hardwood species, mixed species plantations of rainforest hardwoods, conifers and eucalypts, and restoration plantings of a diverse range of rainforest trees found that average above-ground biomass carbon sequestration was greater in restoration plantings than in either mixed species timber plantations or monoculture plantations of native conifers (7.6, 6.1 and 4.8 Mg C ha⁻¹ yr⁻¹, respectively) (Kanowski and Catterall, 2010).

Fast growing tree species (both native and introduced), such as those used in plantations managed for pulpwood, sequester carbon rapidly in above-ground biomass, particularly in the first 5 to 10 years, with typical values for above-ground biomass carbon ranging between 10-20 Mg C ha⁻¹ yr⁻¹ (Brown et al., 1986). These initial rates of *carbon storage* in biomass are usually much greater than in naturally regenerating tropical secondary forests of similar age on former agricultural sites and are directly related to the speed of tree growth and management methods used (Silver et al., 2000; Russell et al., 2010). In longer-rotation (usually 15-50+ years) timber plantations of introduced or native species, carbon sequestration rates in biomass, dead

wood and litter tend to be greater at later ages for slower-growing, high value tree species (Silver et al., 2004). Progressive increases in soil carbon pools occur over time, typically reaching much higher stocks than in pasture or abandoned croplands on which they are established (Silver et al., 2000; Paul et al., 2002; Lal, 2005). Ecosystem carbon sequestration may be increased in mixed-species plantations relative to plantation monocultures if species' mixes involve complementary resource use (i.e., stratified canopy structures) and/or facilitation of tree growth of one species by the other (Piotto et al., 2003; Forrester et al., 2005; Kelt, 2006; see also Section 2.2). There is some evidence that plantation monocultures may also be more vulnerable to stresses (pests, fire, climate change) than more diverse planted forests (Brockhoff et al., 2008), and therefore more likely to become a net *carbon source* in the event of major disturbance (Harris et al., 2006; SCBD, 2011). The extent of use of plantation timber will influence its contribution to carbon accumulation in harvested wood products and to its role in substituting emissions-intensive materials such as steel and concrete.

Enhancing tree cover will generally yield net carbon gains. However, biodiversity benefits will be much more dependent on prior land use and condition, choice of tree species, management practices employed, location, duration of the effort and overall investment (e.g.: Montagnini, 2005; Lamb, 2011). Planted forests, particularly those established for production purposes, usually support reduced biodiversity when compared to primary forests or mature secondary forests, or when such plantations replace non-forest ecosystems, such as grasslands, savannahs and shrublands, as may occur with afforestation (Stephens and Wagner, 2007). When established on highly degraded sites, planted forests can increase biodiversity, particularly where mixed, native species are used (Brockhoff et al., 2008). Tree species' selection is also critical in stimulating ecological succession and thereby creating habitats for a diversity of species (Parrotta et al., 1997; Brockhoff et al., 2008). The planting or restoration of both native and introduced species can enhance biodiversity at the site level by improving soil structure and fertility (Montagnini, 2005; Whisenant, 2005; Parrotta, 1999). Low intensity management practices that minimise soil disturbances and favour retention of naturally regenerating understorey vegetation can create suitable micro-climatic conditions and habitat for indigenous plant (including tree) species and fauna (Parrotta et al., 1997; Lamb et al., 2005; Carnus et al., 2006).

Plantation design and management practices such as tree spacing, thinning, fertiliser and pesticide use will have implications for the diversity of species colonising a restored/reforested site (Lamb et al., 2005; Montagnini, 2005; Holz and Placci, 2005). The use of fertilisers and pesticides can have negative off-site impacts on aquatic biodiversity and on some forest animals (Lindenmayer and Franklin, 2002). Longer rotation plantations, particularly those involving indigenous species, are generally more favourable to biodiversity at the site level compared to short-rotation systems (Lugo, 1992; Keenan et al., 1997; Silver et al., 2004).

There is evidence that when planted, introduced tree species tend to underperform their native counterparts (de Groot and van der Meer, 2011) in delivering a variety of provisioning, regulating and cultural services. Under some conditions, however, introduced tree species can play an effective role during early stages of *forest rehabilitation* (Parrotta et al., 1997; Ewel and Putz, 2004). Negative impacts of introduced tree plantations on biodiversity can be mitigated to some degree by maintaining corridors of native vegetation between single-species plantation blocks. Planting blocks of different tree species in a spatially heterogeneous fashion and thinning to promote structural complexity also enhance biodiversity outcomes (Lamb 1998; Carnus et al., 2006; Brockhoff et al., 2008).

Within a given landscape the spatial scale and location of reforestation and restoration activities are determining factors for biodiversity outcomes. There is a higher potential for rapid re-colonisation of a site and enhanced biodiversity in a landscape with a diversity of native forest species, including seed-dispersing wildlife (Wunderle, 1997; Tucker and Murphy, 1997). While small scale plantings can yield only locally-limited biodiversity benefits, particularly when distant from native forest stands, larger scale (landscape) restoration efforts are more likely to provide more diverse habitats, and enhance provision of a broader array of ecosystem services, including carbon sequestration. Large scale plantation monocultures in areas of high biodiversity might act as a barrier to species' movements, fragmenting populations and reducing genetic diversity, although in some cases, large-scale plantations in degraded landscapes may help to reduce soil erosion and increase soil fertility, and forest restoration along streams and rivers may make a significant difference to local water quality (Lamb et al., 2005; Lamb, 1998). Restoring forests near a protected area or an area of biodiversity importance can significantly increase conservation benefits, notably by reducing *edge effects* (Brockhoff et al., 2008; Lamb et al., 2005). In contrast, establishing intensively-managed plantations of introduced species near biologically-sensitive areas can have major negative outcomes for biodiversity (notably because of pesticide use, introduction of *invasive alien species*, poor habitat quality etc.) (Carnus et al., 2006). Yet, in some cases even monoculture plantations can be considered preferable to alternative land uses - such as intensive agriculture - near a protected area as they can provide a buffer and mitigate other human disturbances (Brockhoff et al., 2008).

3.2.5 Landscape scale planning

Biodiversity considerations can be readily incorporated into national REDD+ planning frameworks using information on the spatial distribution of biodiversity and its threats, as well as known responses of species (or species groups) to different forms of forest disturbance and management.

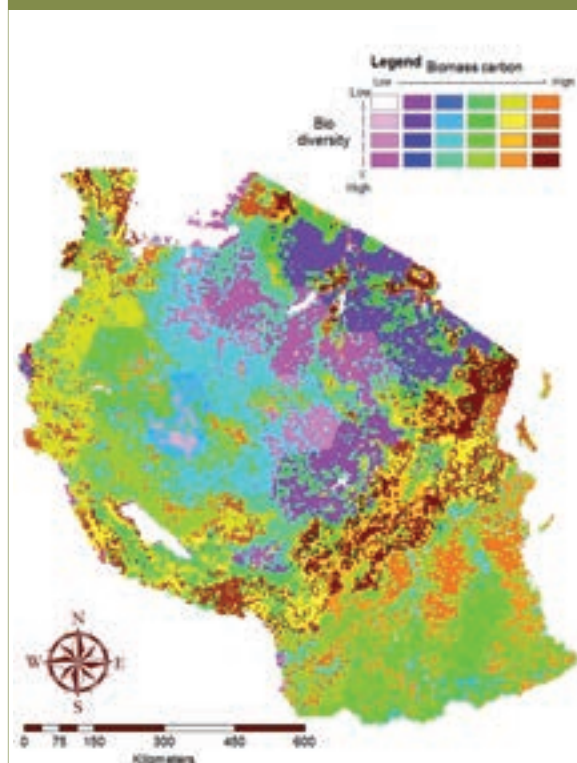
Biodiversity poses a particular challenge for land-use planning and management because the composition of species and habitat types can vary greatly from place to place. Unlike carbon, for which stocks and fluxes can

simply be integrated over space and time, assessing biodiversity depends on both site-level indicators and their spatial and temporal context. Spatial data on the distribution of biodiversity and threats impacting on it are therefore vital in helping to identify priorities for conservation investments that can be compared against spatial priorities for carbon investment (whilst also considering other social, economic and political factors – see Chapter 4) (Gardner et al., 2012). Spatial carbon-biodiversity overlay analyses can be conducted at various scales (where possible incorporating cost data as well) to identify either carbon-neutral solutions that offer varying additional benefits for biodiversity, or high return-on-investment opportunities where relatively minor adjustments to primary carbon objectives can deliver disproportionate benefits for biodiversity (Venter et al., 2009). Analyses can range from a very simple visual comparison of lookup tables of the ecological distinctiveness of different forest

types to spatially explicit optimisation modelling within GIS environments (Wilson et al., 2010). All approaches require spatial carbon and biodiversity information to make more informed choices within national REDD+ programmes (Wendland et al., 2010). Figure 3.2 illustrates one such map for Tanzania, showing how carbon and biodiversity concerns can be effectively illustrated on the same map, potentially identifying regions where the conservation of forest carbon stocks would also increase returns for the conservation of forest mammals (Khan, 2011). Spatial analyses such as these should ideally employ the best biodiversity and threat data that are available, without embarking on costly new field surveys (also see Box 3.2.). A common preoccupation regarding the incorporation of biodiversity concerns into national REDD+ planning relates to the opportunity and management costs that are likely to come from any adjustments to the spatial priorities of REDD+ programmes that are otherwise concerned exclusively with carbon (e.g. Fisher et al., 2011). Nevertheless, one of the most powerful arguments for climate-biodiversity co-financing initiatives is the observation that the trade-off curve between carbon and biodiversity values is non-linear, such that it may be possible to achieve significant improvements (as well as cost-savings) in biodiversity returns while incurring only relatively small carbon penalties (Venter et al., 2009).

Congruence of biomass carbon and mammal species diversity in Tanzania

Figure 3.2



Example national scale map for Tanzania displaying congruence values between carbon and biodiversity at the scale of a 5 km grid and across all vegetation types. Map generated using freely available land cover data from MODIS, mammal data from the freely available African mammal databank (African Mammals Databank (AMD)) and African carbon data provided by UNEP-WCMC, based on multiple sources (Khan, 2011). Such a simple overlay map can help in identifying those areas of both high opportunity (strong positive correlation in carbon and biodiversity values) and risk (low in carbon but high in biodiversity) in the REDD+ planning process.

Sources of spatial data for biodiversity

Box 3.2

Where country-specific spatial data on biodiversity are not available, standardised global data sets can be employed, including maps of globally consistent biogeographical regions (e.g. WWF's ecoregions), areas of particular importance for conservation identified at different scales (e.g. Endemic Bird Areas, Biodiversity Hotspots, Global 200 ecoregions (large areas), Alliance for Zero Extinction Sites and Key Biodiversity Areas (smaller areas) - Schmitt, 2011), and systematically mapped species distribution data (e.g. NatureServe, IUCN Red List of Threatened Species and species group-specific geographic databases such as Herpnet and Antweb). In some parts of the world, region-wide collaborative efforts are emerging to document information on the distribution and threat status of certain species groups, such as the ASEAN Biodiversity Information Sharing Service. To aid analyses of such data, a number of free online tools are being developed to allow coarse-scale analyses that integrate information on biodiversity, carbon and costs to help identify high priorities for REDD+ investments (e.g. InVEST and Marxan). A comprehensive review of currently available biodiversity and forest degradation data, and observational systems has been compiled by the Group on Earth Observation Biodiversity Observation Network (GEO BON, 2011).

Websites:

ASEAN Biodiversity Information Sharing Service - <http://bim.aseanbiodiversity.org/biss/>
InVEST - <http://www.naturalcapitalproject.org/InVEST.html>
Marxan - <http://www.uq.edu.au/marxan/>

Moreover, with finite resources for REDD+ financing, identifying and prioritising regions in which both carbon and biodiversity objectives can be achieved over those in which the focus is only on carbon objectives, will lead to improved overall environmental benefits of REDD+ programmes (as well as potentially securing co-financing from the biodiversity conservation sector).

3.3 Balancing opportunities and risks of different management actions for carbon and biodiversity

There still are large knowledge gaps regarding the longer term impacts of the different REDD+ actions on biodiversity and carbon. Carbon dynamics following deforestation, degradation or forest management are better understood than changes in biodiversity. Management impacts on changes in carbon stocks and fluxes can easily be expressed in common units; they can be summed across spatial scales and integrated over time. By contrast, indicators of impacts on biodiversity are more variable, more complex, depend on spatial and temporal context, and cannot easily be synthesised into single measures or common units. Furthermore, positive or negative biodiversity impacts tend to lag temporally behind carbon impacts and may be more difficult to detect, in the short term, than carbon impacts. Thus, carbon management can be designed more simply to minimise emissions, while biodiversity management is more complex and needs to consider differences among forest types, including their inherent differences in species diversity, and their spatial distributions. Negative impacts on biodiversity (e.g., on native species richness, species composition and ecosystem functioning) can in the long run also have negative effects on carbon sequestration (see Chapter 2).

Impacts on biodiversity and carbon may be correlated (e.g., establishing protected areas to prevent deforestation provides clear benefits for both), but this is not always the case (e.g., plantation forests where increases in carbon stocks can be achieved through management techniques that do not necessarily increase biodiversity). It is invariably much more effective to avert the loss of existing carbon stocks and biodiversity than to restore cleared or degraded areas. Therefore, the largest positive effects for both carbon and biodiversity are associated with actions that reduce the ongoing loss of intact or relatively undisturbed forests. If such forests do not exist in a given area (or deforestation is not an immediate threat), intermediate benefits for both carbon and biodiversity can potentially be obtained by reducing the impacts of extractive uses in managed forest areas that have already been exploited and/or degraded in some way. In these cases, the magnitude of the positive effects on carbon and biodiversity depends on the previous condition of the forest and the degree to which management changes. Even in areas where relatively intact forests are under some future threat (from clearing or conventional logging) the introduction of RIL or other low-impact extractive uses needs to be assessed with caution, as any new economic activity in an area will



Traditional integrated rural land use around rice paddies near Antananarivo, Madagascar.

Photo © Coert Geldenhuys (courtesy of Springer)

always lead to an increased human presence with possible unintended consequences (e.g. increased hunting pressure, over-harvesting and unplanned settlements, or fires).

Given high background levels of threat, forest conservation and efforts to reduce levels of unsustainable extractive uses will deliver significant on-site benefits for carbon and biodiversity, but may still involve substantial socio-economic trade-offs (see Chapter 4). In addition, the overall landscape effects of these actions can result in net negative impacts if extractive activities and other land use are displaced to other areas, i.e., through *leakage* and indirect land-use change (ILUC; Miles and Kapos, 2008; Miles and Dickson, 2010). While UNFCCC requirements for member countries to address national-scale leakage can help to limit the carbon impacts of ILUC, the biodiversity impacts can be considerable, as many low carbon ecosystems can be of high importance for biodiversity. Furthermore, cross-border leakage is a major problem in some areas (Gan and McCarl, 2007). For example, about 39 percent of the significant recovery in forest cover in Vietnam between 1987 and 2006 appears to have been balanced by forest loss directly displaced to nearby countries including Lao PDR, Cambodia and Indonesia (Meyfroidt and Lambin, 2009).

In contrast to concerns about ILUC, there is often an expectation of positive landscape-scale effects from agricultural intensification in farmland to relieve pressures on remaining areas of natural forest. However, in addition to the potential for significant local detrimental impacts of intensified farming (see Section 3.2.1.1), the landscape-scale effects of agricultural intensification are difficult to control and may lead to increases in forest exploitation by attracting more people to an area (e.g., Bertzky et al., 2011; also see Chapter 4, Section 4.4.1)

Restoration, reforestation and afforestation can achieve rapid increases in carbon stocks in degraded areas, although the anticipated site-level contribution to changes in overall carbon stocks is substantially lower than what can be achieved through avoided degradation from conventional logging or avoided deforestation. Positive carbon impacts of reforestation activities can be accompanied by highly variable biodiversity impacts depending on prior land use, extent of degradation and overall landscape condition, choice of planted species, size and location of plantings or other restoration activities, and management practices (see Section 3.2.4). For example, planting introduced species, using monocultures and/or

replacing natural non-forest ecosystems through afforestation can have severe adverse impacts on biodiversity; whereas restoration using native species in degraded sites can bring important biodiversity gains.

Improved fire management can yield significant increases in carbon stocks (through avoided degradation), reductions in emissions of non-CO₂ greenhouse gases and biodiversity benefits in tropical rainforests (Cochrane, 2003; Barlow et al., 2012) and, in ecosystems that naturally burn, such as some of the (sub-) tropical dry forests and woody savannahs (Ryan and Williams, 2011). However, poor fire management (e.g. through long periods of fire suppression) can lead to negative impacts, such as

Table
3.3

State of knowledge on the carbon and biodiversity impacts of REDD+ -related management actions in major forest types. This is based on a broad review of the literature and is intended to highlight knowledge gaps rather than to be a definitive assessment of all available information on all types of management action and all forest types.

Letters in each cell indicate current knowledge of impacts on carbon (**C** = substantial, **c** = some, **0** = lacking) and biodiversity (**B** = substantial, **b** = some, **0** = lacking). Background colour reflects a judgement of the likely incidence of a particular intervention as part of REDD+ action in a given forest type, based on current usage and potential applicability of these interventions, with lighter cells indicating probable lower application of the approach. Those cells of the table where higher likelihood of application is combined with limited knowledge are high priorities for further investigation.

Forest Management Type & Management actions	Existing knowledge of impacts				
	Tropical rain-forest	Tropical (& subtropical) dry forests (& savannahs)	Tropical (& subtropical) montane forests	Tropical (& subtropical) mangrove forests	Tropical (& subtropical) freshwater swamp & peat forests
Improving agricultural practice					
Sustainable agricultural intensification	c, b	c, b	c, b	0, 0	0, 0
Agroforestry	c, b	c, b	c, b	0, 0	0, 0
Sustainable shifting cultivation	C, b	c, b	c, b	0, 0	0, 0
Fire management	C, b	c, b	c, b	0, 0	c, b
Protection Measures					
Strictly Protected areas	C, b	c, b	c, b	c, b	c, b
Multiple use protected areas	c, b	c, b	c, b	0, b	0, 0
Reducing impacts of extractive use					
Reduced Impact Logging	C, B	0, 0	c, b	0, 0	c, b
NTFPs (excluding fuelwood)	0, b	0, b	0, b	0, b	0, 0
Fuelwood	c, 0	C, 0	c, 0	C, b	0, 0
Hunting regulation	c, b	0, b	0, b	0, b	0, b
Restoration/Reforestation					
Enrichment planting	c, b	c, b	c, b	0, 0	0, 0
Assisted natural regeneration	c, b	c, b	c, b	c, b	0, 0
Afforestation & reforestation primarily for wood/fibre production	C, b	c, b	c, b	0, 0	c, b
Reforestation primarily for biodiversity and ecosystem services	c, b	c, b	c, b	c, b	0, 0
Landscape-scale planning & coordination	c, b	c, b	c, b	c, b	c, b

loss of characteristic species, increase in *invasive species* and changes in the water table (Midgley et al., 2010; Jackson et al., 2005), as well as the accumulation of fuel loads in the form of dead wood and leaf litter that can lead to intense fires capable of causing catastrophic environmental damage (Ryan and Williams, 2011).

In addition to the above considerations, implementation of different combinations of REDD+ management actions should, as much as possible, be informed by evidence on their impacts on carbon and biodiversity in different forest types. Amongst the REDD+-relevant forest types introduced in Chapter 2, the greatest body of information on management impacts is available for tropical rainforests, which in general have the highest carbon densities and species richness. Less information is available on other forest types (Table 3.3), making generalisations difficult. Decisions on REDD+ actions need to be taken separately for each forest type, e.g., reforestation of a rainforest area cannot compensate for the loss of an intact dry forest area with lower carbon density and different species diversity.

Landscape scale planning can help to ensure that REDD+ investments effectively consider biodiversity risks and potential gains, as well as potential socio-economic and land-use implications. Ideally, spatial planning processes are supported by general conceptual models that evaluate the potential direct and indirect impacts of REDD+ actions across multiple spatial scales, as well as socio-economic and land-use change implications of particular choices (see Chapters 4 and 5).

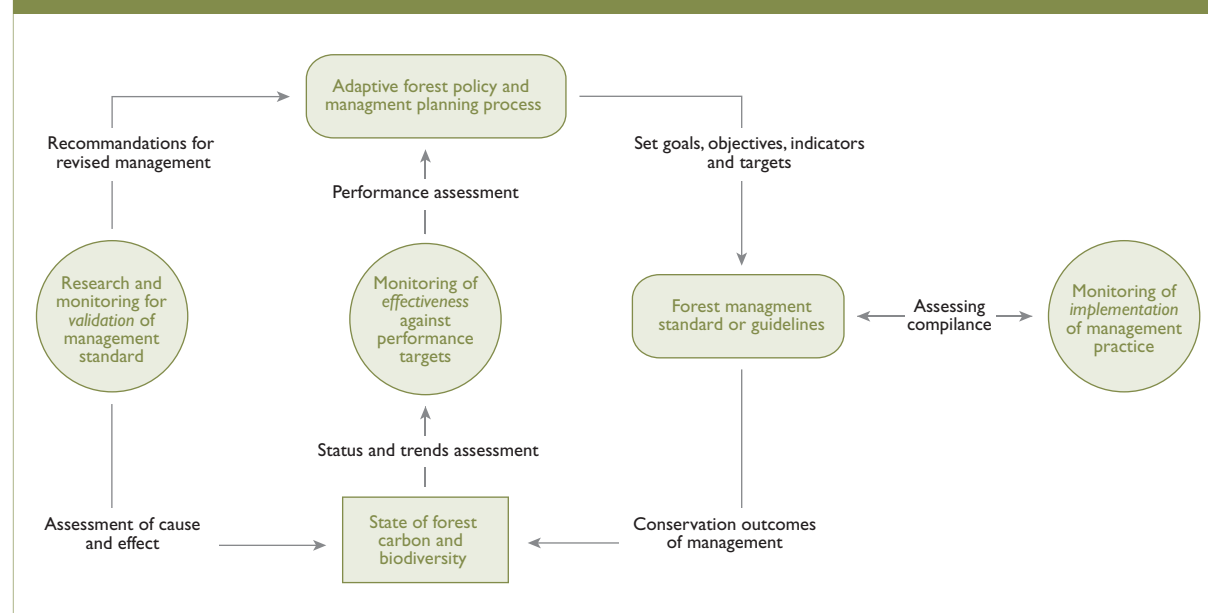
3.4 Monitoring to assess changes in forest carbon and biodiversity

Monitoring is necessary to assess and improve the performance of REDD+ investments in protecting and enhancing carbon and environmental co-benefits. A number of standards, principles and guidance documents have been developed, notably the Cancun safeguards (in Decision 1/CP.16), to measure the application of safeguards in REDD+ activities (see Chapter 5 for a more detailed discussion on these).

Effective monitoring can link ongoing management process and management goals (e.g. as laid out in the objectives of a national, international or third-party management standard or environmental safeguards of REDD+ funding). Irrespective of the type of intervention being used to enhance and/or conserve forest carbon and biodiversity through REDD+ investments, monitoring efforts should go beyond a simple surveillance role, and strive to perform at least three interrelated functions (Gardner, 2010; Figure 3.3). Implementation monitoring assesses whether management practices are being implemented on the ground. Effectiveness monitoring ensures that the implementation of management guidelines translates into minimum levels of performance (i.e. recovery of carbon stocks, maintenance of biodiversity). Validation monitoring evaluates the extent to which existing management standards are adequate and how they can be refined to ensure improvements in management practices towards long-term carbon and biodiversity conservation goals. Effectiveness monitoring satisfies the basic requirements of assessing changes in the status of carbon stocks and

A conceptual framework of an integrated biodiversity monitoring programme for adaptive forest management. To be effective in both assessing and evaluating performance a monitoring programme should comprise three tiers: implementation monitoring of management practice compliance, effectiveness monitoring of the system state against performance indicators, and validation monitoring to evaluate how best to achieve continued progress towards management goals. Figure reproduced from Gardner (2010)

Figure 3.3



biodiversity in a given management area, reporting on stock changes and equivalent emissions reductions, and alerting managers to changes in biodiversity and forest condition. Assessments of the significance of changes in carbon and biodiversity need to be made in comparison to a reference scenario (i.e., without REDD+ intervention).

Indicators to monitor changes in carbon and biodiversity should be: informative with regard to the valued outcomes, e.g. net carbon emissions to the atmosphere and maintenance of forest biodiversity; responsive to management actions; quantitative and measurable; and cost-effective and efficient to collect.

With limited resources and capacity it is sensible to start out with simple approaches based on coarse-scale and/or remotely sensed information before developing more direct field-based assessments (see Section 3.4.3).

3.4.1 Indicators of change in carbon

Changes in the carbon balance of complex landscapes involving different kinds of forest and other land uses cannot be measured directly. Eddy covariance flux towers that directly measure exchanges of CO₂ between the forest canopy and the atmosphere are expensive to operate, require considerable expertise and provide only incomplete time series of measurements that require gap filling when measurements are not possible. Moreover, flux towers typically do not operate during periods of intensive disturbances when major carbon emissions can occur.

Detailed methods to calculate the net balance of carbon emissions and removals are described in the Intergovernmental Panel on Climate Change (IPCC) Good Practice Guidance (IPCC, 2003) and the 2006 IPCC Inventory Guidelines (IPCC, 2006). These methods involve two general approaches: (i) estimation of landscape-level carbon stocks at two points in time, and inference of the net balance of emissions and removals from the difference in carbon stocks (the ‘stock change approach’ - IPCC, 2003), or (ii) the calculation of landscape-scale fluxes associated with tree growth, mortality, decomposition, and natural disturbances and human activities (the ‘default approach’ - IPCC, 2003). The choice of method depends largely on national circumstances, available data and the extent of monitoring infrastructure (see Section 3.4.3).

Implementation of the stock change approach requires information on the area from remote sensing (IPCC, 2003; IPCC, 2006; Sanchez-Azofeifa et al., 2009; Goetz and Dubayah, 2011; GOF-C-GOLD, 2009) and average carbon density (i.e. carbon per hectare) from field measurements (ground plots, forest inventories). Changes in area can be quantified through repeated remote sensing measurements, and periodic rates of change are reported using land-use change matrices that describe the net transition of areas among land classes during the observation period (IPCC, 2003). The advantage of the stock change approach is that it can be easier to implement than the default approach, but the disadvantage is that ancillary data are required to quantify the contribution of non-CO₂ greenhouse gases.

In addition to indicators of forest area and carbon density (by forest type and time since disturbance), assessing the impacts of forest management on the conservation of carbon also requires estimates of ‘productive capacity’, i.e., the quantity of carbon provided annually by the managed landscape in the form of timber, fibre, energy and non-timber forest products. Estimates of productive capacity can be used to distinguish alternate management scenarios for forest landscapes. A forest landscape subject to non-extractive conservation management can store large quantities of carbon but, depending on the age-class structure and degree of past disturbance, such a landscape may be at or near its maximum carbon density, and it does not supply timber, fibre or energy to society, although it provides other productive, cultural and recreational services. The analysis of the fate of harvested carbon and the storage of carbon in harvested wood products (HWP) and landfills (Earles et al., 2012) can also affect the ranking of REDD-related *mitigation* options. In addition to carbon storage in HWP, timber and other woody biomass can provide services that would otherwise have to be supplied using more emissions-intensive materials such as steel, concrete or plastics. Woody biomass can also provide energy that can substitute fossil fuel use.

Using biomass to substitute fossil fuels (which typically have a higher energy density per unit of emission) always results in net increases in atmospheric emissions in the short term (Marland and Schlamadinger, 1995; Manomet, 2010; Searchinger et al., 2009) but these can be offset over time through carbon sequestration in regrowing vegetation. *Displacement factors*, i.e. the amount of greenhouse gas emission reduction per unit of biomass carbon use (Sathre and O’Connor, 2010) are maximised if: (1) the conversion of harvested biomass to end products minimises waste, (2) the end products are used to substitute other emissions-intensive materials, e.g. steel or concrete in building construction, and (3) the end products are used in a cascading system that emphasises reuse, recycling and responsible use of wood products.

Changes in carbon indicators in REDD+ analyses are always assessed relative to a business-as-usual baseline or reference scenario. All three carbon-related indicators - area, carbon stock density and productive capacity - can be integrated over space and time to obtain sub-national or national totals. The carbon-related indicators are relatively straightforward to measure and measurement, reporting and verification (MRV) systems are under development to provide information relevant to the assessment of these indicators, as discussed further below (see Section 3.4.3).

3.4.2 Indicators of change in biodiversity

Assessing changes in biodiversity is much more challenging than assessing changes in carbon because: (i) biodiversity is a broad concept which includes diversity within species, between species and of ecosystems, (ii) spatial differences in biodiversity and associated differences in values to people signify that losses and gains in different areas cannot readily be substituted, and (iii) understanding

of the likely long-term impact of forest management on biodiversity is limited. However, it is possible to identify measurements that can provide ecologically meaningful estimates of change for at least a subset of biodiversity components.

Indicators of biodiversity are commonly divided into structural and compositional measurements - both of which can be assessed at local and landscape scales, and quantified with respect to the total amount (e.g. area of a particular habitat type, number of species) and condition (measured as deviation from a more desirable reference condition) (Gardner, 2010).

The specific indicators most appropriate for a given monitoring project will depend on available resources, management interventions, local expertise, and local and regional values for biodiversity. That said, a number of general recommendations can be made for a common set of basic measurements.

As a minimum requirement biodiversity monitoring should focus on collecting data on structural changes to the forest, at both landscape and local scales (Lindenmayer et al., 2000; McElhinney et al., 2005; Newton, 2007; Gardner, 2010). Indicators of forest structure are the easiest to measure, and provide the most reliable and direct assessment of management impacts and performance.

Landscape-scale measures of forest structure can often be obtained using only remotely-sensed data (Chambers et al., 2007; DeFries, 2008). Numerous studies have demonstrated that the total extent of remaining native habitat is by far the most important factor in determining the biodiversity value of a human-modified landscape (Bennett and Radford, 2007; Lindenmayer et al., 2008; Gardner et al., 2009). There are many ways to assess differences in forest types and levels of fragmentation (Bennett and Radford, 2007; Newton, 2007; Banks-Leite et al., 2011). Simple measures of change in forest area can be enhanced with coarse-scale information on differences in forest type, and analyses of freely available remote sensing data can provide proxy measures of forest degradation from fire and logging. Various measures of landscape configuration are highly correlated, but a simple index of the total amount of forest edge or core forest area (the area of forest more than a minimum distance from an edge) can usually capture most heterogeneity.

Local scale measurements of forest structure can provide valuable information on habitat complexity and resource availability. Specific indicators depend upon the type of impact or management intervention under study. However, many biodiversity-relevant data on forest structure can be captured from standardised forest plot inventories for carbon monitoring, including stem density, basal area and above-ground biomass. Additional stand-level indicators that are worth collecting include measures of canopy cover, understory complexity and dead wood volume.

A direct measure of species-level composition can improve understanding of changes in biodiversity following management interventions. Good ecological disturbance indicators should exhibit a clear response to management interventions at scales that are relevant

to management. In general, the choice of appropriate indicator species should exclude those that are extremely sensitive to disturbance (as they quickly disappear) or exhibit significant lag-periods in their responses to disturbance, those that are very resilient (as they can be found almost everywhere), those that are rare and those for which there are no standardised sampling methods. Insects, herbaceous plants and small vertebrate groups that are closely dependent on local habitat changes can provide valuable information on stand-level impacts, whereas more mobile taxa such as bats, many birds and mammals, and fish whose persistence depends upon connectivity over large areas, can provide complementary information on landscape level forest changes. Birds are a commonly favoured ecological disturbance indicator group for biodiversity monitoring (Bibby, 1999) as they have been shown to respond to environmental changes over many scales (Cushman and McGarigal, 2002). Given the diversity of impacts that human activities exert on natural systems, it is advisable, to employ a number of different indicator groups that reflect different levels of biological organisation, and different spatial scales and types of management impact (Noss, 1990; Angermeier and Karr, 1994).

Additional species indicators may be necessary to assess the conservation status and requirements of individual species that either play particularly important functional roles in the forest ecosystem, are highly threatened, or are of particular economic or cultural value to stakeholders (Lindenmayer et al., 2007; Gardner, 2010). Inclusion of species, such as those of particular conservation concern or societal importance may be required by law or voluntary management standards to be included in a biodiversity monitoring programme.

Interpretation of observed changes in species indicators needs to be based on a basic conceptual framework of cause and effect - whereby human activities can be clearly linked to changes in forest structure, which in turn can be associated with changes in the amount and composition of biodiversity (Guynn et al., 2004; Niemi and McDonald, 2004; Niemeijer and de Groot, 2008; Gardner, 2010).

3.4.3 Putting field carbon and biodiversity monitoring into practice

The collection of both carbon and biodiversity indicator data from the field is a costly and time consuming undertaking, yet the efficient collection of appropriate data at a carefully selected sub-sample of sites is important for validating indicators based on remote-sensing data only. The sub-sample of sites within projects selected for biodiversity monitoring should be targeted towards areas of forest that are undergoing the greatest changes (whether through clearance, degradation or restoration) so that monitoring data can help improve estimates of biodiversity responses to REDD+ activities.

Effective integration of biodiversity considerations into REDD+ MRV systems is essential if they are to be viable and neither overburden national capacity nor be ignored because they are too costly to implement.

The IPCC has defined a three tiered approach to carbon emissions assessments in its guidelines², with different tiers requiring increasingly complex data and analyses to reduce uncertainties in estimates (IPCC, 2006). The tiered approach to assessment and monitoring enables countries to assess and report on emissions even when national data and capacity are limited, and outlines how improvements can be achieved. It provides a clear structure for promoting transparency, consistency and accuracy.

In an analogous way to carbon MRV, it is possible to identify tiers of data requirements and analytical complexity for biodiversity assessments based on the indicators - whereby remote-sensed landscape-scale indicators of forest structure (spatial extent of different forest types and fragmentation) represent the lowest tier, and species-level field monitoring of changes in the diversity and relative abundance indicator groups represents the highest tier (Gardner et al., 2012). Irrespective of the type of indicators used for monitoring, efforts to conserve and restore forest carbon and biodiversity will take place at multiple spatial scales and within different governance contexts (also see Chapter 4, Sections 4.2.1 and 4.5), with individual projects nested within sub-national initiatives and national planning frameworks (Chagas et al., 2011). A major challenge is to ensure that data collected for individual projects can be (at least partially) scaled up to help meet reporting requirements and assess performance at larger spatial scales. This may be addressed in two ways. First, field data on changes in carbon and biodiversity following management impacts can be used to assess the adequacy of, and potential for improvement in, forest management standards for different types of intervention. The extent to which good practice standards are then implemented in other areas can thereby provide a valuable indicator of forest management performance at regional or national scales (even if field monitoring of carbon and biodiversity is not conducted at every site). The second approach to scaling up is through the validation of remote-sensing data on landscape structure and forest structural degradation (Gardner, 2010; FAO, 2011; Herold et al., 2011) with plot-based samples of above-ground biomass and biodiversity.

Determining who should be responsible for designing and implementing carbon and biodiversity monitoring programmes depends on both the level of detail required as well as the people and institutions which the data are intended to benefit. In many places an integrated approach to monitoring that combines guidance and management from scientific and technical experts with close involvement of local people (i.e., forest managers and representatives of affected local communities) may be optimal (Danielsen et al., 2009; Gardner, 2010). The involvement of local people in the design and implementation of monitoring programmes can empower those who are ultimately responsible for management, and provide

a cost-effective and sustainable means of data collection and a potentially rich source of local knowledge to aid interpretation of results (Danielsen et al., 2010).

Integrating the collection of biodiversity data with the collection of carbon stock data from the same set of forest monitoring plots that are required under Tier 3 of the IPCC system (Teobaldelli et al., 2010) will reduce costs and ensure the effective involvement of local people (Danielsen et al., 2010). If designed appropriately (i.e. stratified towards areas of greatest forest change), pre-existing National Forest Inventory plots may be suitable for this task.

3.5 Conclusions

1. Overall, outcomes of REDD+ actions are likely to bring positive impacts for both carbon and biodiversity. Actions that seek to maintain existing carbon and biodiversity through effectively reducing deforestation and forest degradation are more likely to have the greatest and most immediate benefits for carbon and biodiversity compared to those that seek to restore them. Strategic spatial planning of REDD+ actions can potentially deliver major benefits for biodiversity.
2. Different REDD+ actions can have highly variable impacts on carbon and biodiversity, depending on location, scale of implementation, initial conditions, historical impacts, forest type and the wider landscape context. Different actions require different time periods to deliver benefits for carbon and biodiversity, with biodiversity benefits in some cases being achieved more slowly than carbon benefits.
3. REDD+ actions may fail to deliver biodiversity benefits and in some cases may cause negative impacts. Trade-offs between carbon and biodiversity outcomes can occur both locally and at wider spatial scales. For example, plantations of introduced species may provide large and rapid carbon benefits while contributing little to local biodiversity, or depending on factors such as their management and prior land uses, may actually have detrimental impacts. At landscape scales, efforts to alleviate deforestation pressure on natural forests through agricultural intensification and associated inputs of agrochemicals can lead both to detrimental impacts on biodiversity and to increased greenhouse gas emissions.
4. Not all impacts on carbon and biodiversity are easily anticipated or measured. Impacts can occur outside the area of management and/or in the future. Indirect impacts resulting from displacement of land use pressures or extractive activities, e.g. following the creation of protected areas, are particularly problematic.

² Tier 1 employs default emissions factors (biomass estimates from different ecoregions) from the IPCC Guidelines (IPCC, 2006), Tier 2 includes country-level emission factors for regional forest strata and explicit consideration of data uncertainties, and Tier 3 uses actual inventory data and repeated plot measurements to quantify and model changes in individual carbon pools.

Unintended increases in net GHG emissions may result if constraints on timber extraction lead to the replacement of wood products with more emissions-intensive alternatives such as concrete, steel or plastics. Both the magnitude and the direction of the impacts can change over time. For example, fire suppression in naturally fire-dependent forest ecosystems can lead to increased carbon stocks in the short term, but can be severely detrimental in the long term for both carbon and biodiversity if the accumulation of fuel leads to catastrophic fires.

5. Impacts of REDD+ interventions are likely to vary significantly across different forest types and landscape conditions. Therefore, caution is needed in extrapolating management recommendations across different ecosystems, and the development of regionally-tailored strategies for REDD+ remains a major priority for future research.
6. Opportunities exist for using data obtained from the measurement, reporting and verification of carbon outcomes to derive landscape-scale proxies for changes in biodiversity (e.g. changes in the spatial extent and fragmentation of different forest types), but these are not sufficient for a full assessment of biodiversity impacts and trends. More detailed spatial data are needed on patterns of biodiversity, expected trends in forest cover and condition, and existing management actions. These can be used to provide better understanding of the impacts of different REDD+ actions, which is needed to guide integrated planning processes.
7. Existing knowledge is very incomplete, particularly with respect to the biodiversity impacts of REDD+ actions, as well as their indirect effects on forest ecosystems at landscape and regional scales. Nevertheless, current understanding is sufficient to significantly improve efforts to minimise environmental harm and maximise multiple benefits of REDD+ actions.
8. Even interventions that have positive direct impacts on both carbon and biodiversity, such as effective protection of natural forests and forest restoration on deforested lands, may result in negative social and economic impacts (see Chapter 4) or may be constrained by political or governance factors (see Chapter 5).

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