



Understanding Relationships between Biodiversity, Carbon, Forests and People: The Key to Achieving REDD+ Objectives

A Global Assessment Report

Prepared by the Global Forest Expert Panel on Biodiversity, Forest Management and REDD+

Editors: John A. Parrotta, Christoph Wildburger, Stephanie Mansourian

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Preface

This volume is the third assessment report of the Global Forest Expert Panels (GFEP) initiative, following those on adaptation of forests and people to climate change (in 2009), and on international forest governance (in 2011). The mission of GFEP is to support forest-related intergovernmental processes by assessing available scientific information on forest-related issues of high concern in a comprehensive, interdisciplinary, objective, open and transparent way. GFEP is a joint initiative of the Collaborative Partnership on Forests. It is led and coordinated by the International Union of Forest Research Organizations (IUFRO).

Forests harbour a major proportion of the world's terrestrial biodiversity and provide a wide range of vitally important ecosystem services – including carbon sequestration and storage. Deforestation and forest degradation continue to erode biodiversity and the capacity of forest ecosystems to help mitigate climate change and provide the goods and services that sustain livelihoods and human well-being locally, and globally. Reducing greenhouse gas emissions from deforestation and forest degradation, and enhancing forest carbon stocks in developing countries (REDD+) is a proposed mechanism which has the potential to realise its primary objective – climate change mitigation – with variable impacts, positive and negative, on biodiversity, forests and people. REDD+ is complex, its proposed activities and implementation mechanisms not yet clearly defined, and therefore surrounded by uncertainty. Because of its high relevance to climate change mitigation, the conservation and sustainable use of forests and their biological diversity, the Expert Panel on Biodiversity, Forest Management and REDD+ was established by the Collaborative Partnership on Forests in December 2011 to carry out this assessment.

The Expert Panel included 24 scientists and other experts from a variety of biophysical and social science disciplines relevant to the topics covered in this assessment report. An additional 18 contributing authors added their expertise to the assessment. Each chapter was prepared by a team of Lead Authors and Contributing Authors led by one or more Coordinating Lead Authors. A full draft of the report and its individual chapters was peer-reviewed prior to its completion. The results of this voluntary collaboration between January and October 2012 are presented in the six inter-related chapters comprising this book.

This assessment report evaluates the implications of forest and land management interventions envisaged under REDD+ in a multidimensional and integrated fashion. It summarises the most current scientific literature that sheds light on the relationships between forest biodiversity and carbon (and other ecosystem services), how these complex relationships may be affected by management activities implemented to achieve REDD+ objectives, the potential synergies and trade-offs between and among environmental and socio-economic objectives, and their relationship to governance issues. Based on the main findings of the assessment (summarised in Chapter 6), a policy brief entitled 'REDD+, Biodiversity and People: Opportunities and Risks' has been prepared especially for policy- and decision-makers.

Given the broad scope of this assessment, it was not possible to cover all topics in great detail or to the extent that some readers may have wished. However it is my hope that this assessment report provides a sound scientific basis for informed decision-making by policy-makers, investors, donors and other interested stakeholders with respect to REDD+ implementation.



John A. Parrotta

*Chair, Global Forest Expert Panel on Biodiversity,
Forest Management and REDD+*

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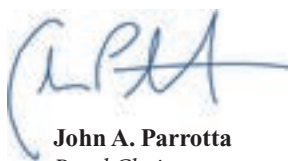
This publication is the product of the collaborative work of scientific experts who served as authors in different capacities: coordinating lead authors, lead authors and contributing authors. We express our sincere gratitude to all of them: Toby Gardner, Valerie Kapos, Werner A. Kurz, Constance L. McDermott, Bernardo B.N. Strassburg, Ian D. Thompson and Bhaskar Vira (coordinating lead authors); Samuel Assembe Mvondo, Joice Ferreira, Manuel Guariguata, Markku Kanninen, Sango Mahanty, Adrian Martin, Kimiko Okabe, Yude Pan, Lian Pin Koh, Nophea Sasaki, Christine B. Schmitt, Charlotte Streck, Jason Tylianakis and Harro van Asselt (lead authors); Jos Barlow, Neil M. Dawson, Amy E. Duchelle, Simone Frick, Nicole Gross-Camp, Constanze Haug, David Humphreys, Agnieszka Latawiec, Marco Lentini, Harisharan Luintel, Mari Mulyani, Cleber Salimon, Chandra Shekhar Silori, Mark D. Spalding, Regan Suzuki, Luke Swainson, Nathalie van Vliet and Fariborz Zelli (contributing authors). Without their voluntary efforts and commitment the preparation of this publication would not have been possible. We are also grateful to the institutions and organisations to which the authors are affiliated for enabling them to contribute their expertise to this assessment.

We acknowledge and sincerely thank the reviewers of the full report and the various chapters whose comments have greatly improved the quality of this publication: Arun Agrawal, Susan Braatz, Jean-Michel Carnus, Bryan Finegan, Alain Franc, Peter Graham, Robert Jandl, Peter Kanowski, Shashi Kant, Katia Karousakis, David Lamb, Ariel E. Lugo, Robert Nasi, Ravi Prabhu, Michael Pregernig and Francis E. Putz.

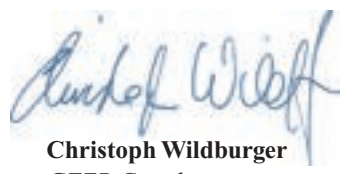
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Furthermore, we would like to thank the member organisations of the GFEP Steering Committee for providing overall guidance and generous in-kind support: the International Union of Forest Research Organizations (IUFRO), the Food and Agriculture Organization of the United Nations (FAO), the Secretariat of the United Nations Forum on Forests (UNFF), the Secretariat of the Convention on Biological Diversity (CBD), the Center for International Forestry Research (CIFOR), and the World Agroforestry Centre (ICRAF). Our special thanks go to the team of the IUFRO Secretariat for providing indispensable administrative and technical support to the work of the Panel. We are particularly grateful to the Food and Agriculture Organization of the United Nations (FAO), the Federal Research and Training Centre for Forests, Natural Hazards and Landscape (BFW, Vienna) and to Embrapa Amazonia Oriental (Belém, Brazil) for hosting Expert Panel meetings.

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Acronyms, units and symbols

AGB	Above-ground Biomass
AHTEG	Ad Hoc Technical Expert Group
AMD	African Mammals Databank
ASEAN	Association of Southeast Asian Nations
B-EF	Biodiversity-Ecosystem Functioning
BFW	Federal Research and Training Centre for Forests, Natural Hazards and Landscape
BGB	Below-ground Biomass
CBD	Convention on Biological Diversity
CCBA	Climate, Community and Biodiversity Alliance
CDM	Clean Development Mechanism
CFS	Canadian Forest Service
CFUG	Community Forest User Group
CIFOR	Center for International Forestry Research
CLUA	Climate and Land Use Alliance
COMIFAC	Central African Forests Commission
COP	Conference of the Parties
CPF	Collaborative Partnership on Forests
DFID	Department for International Development
DRC	Democratic Republic of the Congo
EIA	Environmental Impact Assessment
ES	Ecosystem Service
ESMF	Environmental and Social Management Framework
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FCPF	Forest Carbon Partnership Facility
FECOFUN	Federation of Community Forest Users, Nepal
FIP	Forest Investment Programme
FLEG	Forest Law Enforcement and Governance
FLEGT	Forest Law Enforcement, Governance and Trade
FPIC	Free, Prior and Informed Consent
FSC	Forest Stewardship Council
GDP	Gross Domestic Product
GEF	Global Environment Facility
GEO BON	Group on Earth Observation Biodiversity Observation Network
GFEP	Global Forest Expert Panels
GHG	Greenhouse Gas
GPP	Gross Primary Production
HWP	Harvested Wood Product
ICCN	Institut Congolais pour la Conservation de la Nature
ICRAF	World Agroforestry Centre
ICV	Instituto Centro de Vida
IFAD	International Fund for Agricultural Development
IFF	International Forum on Forests
IFT	Instituto Floresta Tropical
ILUC	Indirect Land-Use Change
IPF	International Panel on Forests
ITTO	International Tropical Timber Organization
IUCN	International Union for Conservation of Nature
IUFRO	International Union of Forest Research Organizations
LATIN	Lembaga Alam Tropika Indonesia
MDG	Millennium Development Goal
MEA	Millennium Ecosystem Assessment

MRV	Monitoring (or Measurement), Reporting and Verification
NBP	Net Biome Production
NBSAP	National Biodiversity Strategy and Action Plan
NEE	Net Ecosystem Exchange
NEP	Net Ecosystem Production
NGO	Non-Governmental Organisation
NPP	Net Primary Production
NTFP	Non-Timber Forest Product
PA	Protected Area
PEFC	Programme for the Endorsement of Forest Certification
PES	Payment for Ecosystem (or Environmental) Service
PFE	Permanent Forest Estate
PFM	Participatory Forest Management
POWPA	Programme of Work on Protected Areas
PRODEMFLOR	Forest Management Development Programme
PSR	Pressure-State-Response
RECOFTC	Regional Community Forestry Training Centre
REDD	Reducing Emissions from Deforestation and Forest Degradation
RIL	Reduced Impact Logging
R-PP	Readiness Preparation Proposal
SBSTTA	Subsidiary Body on Scientific, Technical and Technological Advice
SD	Standard Deviation
SE	Standard Error
SEA	Strategic Environmental Assessment
SEPC	Social and Environmental Principles and Criteria
SES	Social and Environmental Standards
SESA	Strategic Environmental and Social Assessment
SFM	Sustainable Forest Management
SISA	System of Incentives for Environmental Services
TFAP	Tropical Forestry Action Plan
TNC	The Nature Conservancy
UK	United Kingdom of Great Britain and Northern Ireland
UNDRIP	UN Declaration on the Rights of Indigenous Peoples
UNEP-WCMC	United Nations Environment Programme – World Conservation Monitoring Centre
UNFCCC	United Nations Framework Convention on Climate Change
UNFF	United Nations Forum on Forests
UNPFII	UN Permanent Forum on Indigenous Issues
US	United States of America
VCS	Verified Carbon Standards
VPA	Voluntary Partnership Agreement
WWF	Worldwide Fund for Nature

The International System of Units (SI) is used in the publication.

Mg = megagram (1 Mg = 10 ⁶ g)	W = watt
Tg = teragram (1 Tg = 10 ¹² g)	C = carbon
G = giga (10 ⁹)	CO = carbon monoxide
P = peta (10 ¹⁵)	CO₂ = carbon dioxide
ha = hectare (100 ha = 1 km ²)	CH₄ = methane
yr = year	N₂O = nitrous oxide



Chapter I

Introduction

Coordinating lead author: John A. Parrotta

Lead authors: Toby Gardner, Valerie Kapos, Werner A. Kurz, Stephanie Mansourian, Constance L. McDermott, Bernardo B. N. Strassburg, Ian D. Thompson, Bhaskar Vira and Christoph Wildburger

The relationships between *biodiversity*¹, *carbon*, *forests* and people are complex and interdependent. Reducing the rates of global *deforestation* and *forest degradation* will yield substantial gains for *climate change mitigation* and biodiversity conservation. Under appropriate conditions, it could also achieve significant social and economic gains. The degree to which these goals are met through a mechanism such as REDD⁺ will depend on the specific policies and practices employed. Should biodiversity and human well-being not be given sufficient consideration, there is a very real risk that REDD⁺ may fall short in achieving its objectives.

To ensure that benefits from REDD⁺ are achieved, it is important to understand the underlying scientific premises for reducing *emissions* from deforestation and forest degradation; the relationships between carbon, biodiversity and people and how these are affected by management, as well as the broader *governance* context which frames REDD⁺. This assessment report aims to further this understanding by providing recent and policy-relevant scientific information to support decision-making on activities for meeting REDD⁺ objectives.

1.1 Forests, carbon and biodiversity

Covering about a third of the earth's land surface (just over 4 billion hectares – FAO, 2010) forests play a major role in the global carbon cycle and contain a substantial proportion of the world's terrestrial biodiversity. Forests also provide a broad range of other '*ecosystem services*' – the benefits people obtain from *ecosystems*. These ecosystem services include supporting services such as nutrient cycling, soil formation and primary productivity; provisioning services such as food, water, timber and medicine; regulating services such as erosion control, climate regulation, flood mitigation, purification of water and air, pollination and pest and disease control; and cultural services such as recreation, ecotourism, educational

and spiritual values (MA, 2005). Deforestation and forest degradation in the tropics and sub-tropics have a large negative impact on terrestrial biodiversity, and thus on the provision of those ecosystem services that are most closely linked to biodiversity.

One of the key supporting services provided by forests is carbon removal from the atmosphere (*sequestration*³) and the long-term storage of this carbon in *biomass*, dead organic matter and soil carbon pools. Of the global forest *carbon stocks*, an estimated 55 percent (471 Pg C) is stored in (sub-)tropical⁴ forests, of which more than half is stored in biomass (Pan et al., 2011). The role of forests in sequestering carbon is evident when considering that 57 percent of the carbon emitted annually from global fossil fuel use and land-use change is absorbed by land and ocean sinks, cutting in half the rate of increase in atmospheric CO₂ concentrations over the past four decades (Le Quéré et al., 2009). Specifically, forests globally are estimated to have contributed a net sink of 1.1 Pg C yr⁻¹ between 1990 and 2007. In (sub-)tropical regions, while intact forests absorb 1.2 Pg C yr⁻¹, this amount is offset by the net emissions resulting from land-use changes (i.e., deforestation and clearing emissions minus regrowth storage) of 1.3 Pg C yr⁻¹ (Pan et al., 2011), making (sub-)tropical forest regions a net source of atmospheric carbon of approximately 0.1 Pg C yr⁻¹ (Pan et al., 2011). These figures highlight the very fine line between the (sub-)tropical regions acting as a net source of *carbon emissions* or a net *carbon sink*.

Today, more than ever, the future of the global forest carbon sink is highly uncertain. The loss of biodiversity, linked to deforestation and forest degradation, could further diminish the ability of forests to effectively provide multiple ecosystem services, including, carbon sequestration. As a result, human well-being – particularly for those most dependent on forests and most vulnerable – could be significantly and adversely impacted. Equally the loss of biodiversity could further tip the balance leading to (sub-)

¹ Throughout this assessment report, all terms that are defined in the glossary are introduced for the first time in a chapter using *italics*.

² Reducing emissions from deforestation and forest degradation, and enhancing forest carbon stocks in developing countries.

³ Throughout this assessment report, '(sub-)tropical' includes both tropical and sub-tropical regions.

tropical forested regions becoming growing sources of carbon emissions. In this context, efforts to reduce deforestation and forest degradation are of critical value.

1.2 Impacts of deforestation and forest degradation

Deforestation, resulting mainly from conversion of forests to agriculture, has been estimated at between 13 to 16 million hectares (Mha) per year between 1990 and 2010 (FAO, 2010). However, as a result of large-scale forest planting efforts, natural expansion of forests, and successes in slowing deforestation rates in some countries, the net global loss in forest area has slowed from 8.3 million ha (1990 to 2000) to 5.2 Mha (2000-2010) (FAO, 2010). Forest loss is the second largest anthropogenic source of carbon dioxide emissions to the atmosphere, contributing the equivalent of about 12 percent of fossil fuel emissions in 2008 (van der Werf et al., 2009; Pan et al., 2011). Deforestation results in immediate CO₂ emissions (with small amounts of CO, CH₄ and N₂O) when biomass and dead organic matter is burned, and in slower releases when biomass and dead organic matter decay. At the same time, deforestation is the major cause of global biodiversity loss in terrestrial ecosystems (SCBD, 2010). The loss of forest cover and related ecosystem services has a range of negative repercussions on local stakeholders, including the poor and most vulnerable.

Forest degradation - or changes in forest condition that result in the reduction of the capacity of a forest to provide goods and services - also contributes to global anthropogenic CO₂ emissions, as well as reductions in biodiversity. It has been estimated that the area of degraded forests⁴ in tropical regions increased by 2.4 million ha yr⁻¹ during the 1990s (Nabuurs et al., 2007).

The Intergovernmental Panel on Climate Change (IPCC) has stated that forest-related mitigation activities can considerably reduce emissions from sources and increase CO₂ removals by sinks at low costs, and can be designed to create synergies with *adaptation* and sustainable development (IPCC, 2007). Reducing or reversing forest degradation in (sub-)tropical regions will also contribute to climate change mitigation given the significant impacts of forest degradation on global biodiversity and ecosystem services, including carbon sequestration. Yet the means by which reductions in deforestation and forest degradation are accomplished will determine the rate of change, and the extent and type of impacts on forest biodiversity and on the broader range of services provided by forests at local to global scales.

Actions taken to enhance the role of forests in climate change mitigation may have positive, neutral or negative impacts on the capacity of forests to provide specific benefits to society. The REDD+ interventions themselves can have substantial socio-economic consequences, both positive (such as increased financial flows to poor communities) and negative (such as the loss of access to forest

resources). They may also have consequences beyond forests, for example, altering the distribution of and incentives for other forms of land use, including agriculture. Further, if actions to enhance the role of forests in climate change mitigation are to be effective and long-lasting they must adequately address the underlying causes of deforestation and forest degradation, including increased demand for agricultural land, timber and other forest products, lack of inter-sectoral policy coordination and weak governance. The interactions, relationships and potential trade-offs and compromises among mitigation objectives, biodiversity and ecosystem services outcomes, and the needs and aspirations of stakeholders need to be understood, negotiated and reconciled.

1.3 REDD+: A moving target

Reducing emissions from deforestation and forest degradation, and enhancing forest carbon stocks in developing countries (REDD+) is a proposed mechanism for climate change mitigation. It has been designed to encourage developing countries to contribute to climate change mitigation through the following five sets of (non-exclusive) activities: reduction of emissions from deforestation and forest degradation; conservation of forest carbon stocks; sustainable management of forests; and enhancement of forest carbon stocks.

The notion of REDD+ means different things to different countries, organisations and individuals. Its strongest proponents see it as a quick, relatively inexpensive option for mitigating climate change that will mobilise significant resources and successfully achieve its objectives. Many hope that it will also stimulate efforts to transform national policies and governance systems to meet biodiversity conservation goals and improve the livelihoods of people through, for example, more sustainable management of forests and forest landscapes, resolution of long-standing land tenure issues, and improved coordination of policies between forest, agriculture, energy and other sectors. By contrast, the critics of REDD+ hold different views, emphasising a lack of clarity regarding the eventual architecture of the international REDD+ regime and the international financial mechanisms that will underpin it, the environmental and social risks and inequity associated with various aspects of REDD+ policy development, planning and implementation (e.g., issues of sovereignty, risk of 'land grabs') and long-standing difficulties in addressing the underlying causes of deforestation and forest degradation.

The topic of reducing emissions from deforestation and forest degradation in developing countries was first introduced at the eleventh session of the Conference of the Parties (COP) to the United Nations Framework Convention on Climate Change (UNFCCC) in Montreal in December 2005. The 'Bali Action Plan' which emerged from the 13th session of the Conference of the Parties to the UNFCCC in December 2007, acknowledged that

⁴ When defined as a decrease in forest stand density or increase of disturbance in forest classes

reducing *greenhouse gas* emissions from deforestation and forest degradation (i.e., REDD) could potentially yield a range of environmental and social ‘co-benefits’ that could complement the aims and objectives of other multilateral agreements (discussed in Chapter 5 of this report).

Three years later, in December 2010, at the 16th session of the UNFCCC Conference of the Parties in Cancún, Mexico, an agreement was reached on policy approaches and positive incentives on issues relating to reducing greenhouse gas emissions from forests. The Cancún decision on REDD+ (Decision 1/CP.16 paragraph 70) specifically encourages developing country Parties to contribute to mitigation actions in the forest sector by undertaking the following activities, as deemed appropriate by each Party and in accordance with their respective capabilities and national circumstances:

- a) Reducing emissions from deforestation;
- b) Reducing emissions from forest degradation;
- c) Conservation of forest carbon stocks;
- d) Sustainable management of forests;
- e) Enhancement of forest carbon stocks.

These five REDD+ activities should:

- Be country-driven and be considered options available to Parties (Appendix I, para. 1a);
- Be consistent with the objective of environmental integrity and take into account the multiple functions of forests and other ecosystems (Appendix I, para. 1c);
- Be implemented in the context of sustainable development and reducing poverty, while responding to climate change (Appendix I, para. 1g);
- Be consistent with the adaptation needs of the country (Appendix I, para. 1h);
- Be results-based (Appendix I, para. 1j);
- Promote sustainable management of forests (Appendix I, para. 1k).

Of particular relevance to this assessment report, are questions related to impacts of REDD+ activities on biodiversity and forest ecosystem services. It is generally accepted that of the five REDD+ activities, reducing deforestation and forest degradation have by far the greatest potential to yield positive carbon and biodiversity outcomes. As a means to enhance forest carbon stocks, forest *restoration* to create corridors and improve forest connectivity in fragmented landscapes can provide substantial benefits for biodiversity. There is much uncertainty, however, about the potential impacts on biodiversity of other activities to enhance forest carbon stocks and those related to the sustainable management of forests. Further, there is uncertainty and concern about how all REDD+ activities may directly and indirectly affect the well-being of people, especially indigenous and local communities.

Opportunities envisaged through REDD+ include, among others, increased policy support, incentives and financial resources to: improve *in situ* conservation and maintain vital ecosystem services and production *forest management* practices; improve livelihoods and forest governance; and support better monitoring and reporting

of forests and their biodiversity and ecosystem services. On the other hand, some REDD+ activities could negatively affect both biodiversity and people including conversion of forests of high biodiversity value to other types of forest, the *afforestation* (i.e., conversion) of non-forest ecosystems such as grasslands and savannahs, the displacement or disenfranchisement of rural communities, and increased social inequities. Other concerns related to the potential indirect impacts of REDD+ activities on biodiversity include risks of displacement of deforestation and forest degradation to new areas that may have high biodiversity value.



Uapaca sp., Parc National de l'Indo, Gabon
Photo © Robert Nasi

In response to concerns about the potential negative impacts of REDD+ activities on biodiversity and local people, UNFCCC Decision 1/CP.16 (Appendix I) states that the following safeguards should be promoted and supported when undertaking REDD+ activities:

- That actions complement or are consistent with the objectives of national forest programmes and relevant international conventions and agreements (para. 2a);
- Transparent and effective national forest governance structures, taking into account national legislation and sovereignty (para. 2b);
- Respect for the knowledge and rights of indigenous peoples and members of local communities, by taking into account relevant international obligations, national circumstances and laws, and noting that the United Nations General Assembly has adopted the United Nations Declaration on the Rights of Indigenous Peoples (para. 2c);
- The full and effective participation of relevant stakeholders, in particular, indigenous peoples and local communities (para. 2d);

- That actions are consistent with the conservation of natural forests and biological diversity, ensuring that [REDD+] activities are not used for the conversion of natural forests, but are instead used to incentivise the protection and conservation of natural forests and their ecosystem services, and to enhance other social and environmental benefits (para. 2e);
- Actions to address the risks of reversals (para. 2f);
- Actions to reduce displacement of emissions (para. 2g).

At the UNFCCC's 17th Conference of the Parties in Durban, South Africa, in November/December 2011, the COP 16 decision was elaborated and guidance offered on systems for providing information on how environmental and social safeguards related to REDD+ activities are addressed and respected.

In the same decision, UNFCCC Parties also agreed on modalities for reference levels for forests and forest emissions as benchmarks for assessing each country's performance in implementing REDD+ activities. The conferences in Cancun and Durban also explored financing options for the implementation of results-based REDD+ actions, including establishment of the 'Green Climate Fund'. Progress on these issues will be reported at UNFCCC's 18th session of the Conference of the Parties in Doha, Qatar in November/December 2012. Negotiations at this meeting are also expected to identify policy instruments that could address national and international drivers of deforestation and forest degradation (e.g., agriculture), and existing perverse policy incentives.

The evolution of the international REDD+ regime, and development of 'safeguards', is of considerable interest to the Convention on Biological Diversity (CBD), the Food and Agriculture Organization of the United Nations (FAO), the United Nations Forum on Forests (UNFF), other members of the Collaborative Partnership on Forests and to a broad spectrum of other organisations promoting the conservation and sustainable use of biological diversity as well as the rights and interests of indigenous and local communities who may have the most to gain, or lose, from REDD+ implementation. Within the CBD, discussions on the linkages between REDD+ and biodiversity conservation have increased in recent years (as discussed in Chapter 5).

1.4 Purpose and scope of this assessment report

The likelihood of REDD+ activities delivering positive climate mitigation results and social and environmental co-benefits, will hinge on key choices made by decision-makers (policy-makers, investors, planners, land managers and other relevant stakeholders), since the management of forest stands and forest landscapes for net positive carbon benefits will have implications for biodiversity and ecosystem services other than carbon sequestration. These choices, which will inevitably involve trade-offs among land uses and forest-based ecosystem services, and among stakeholders at all levels, need to be understood and integrated into REDD+ decision-making, planning and management processes. They concern, for example, the selection and design of the most appropriate REDD+ activities to be implemented, the scale at which to implement them, objectives of the investors, and the balance between local and international impacts (particularly as they relate to land use and food security). If they are to lead to desired outcomes, these choices should be informed by the best available knowledge regarding the likely impacts (ecological and socio-economic) of REDD+ actions.

1.4.1 Terms of reference

The thematic 'Expert Panel on Biodiversity, Forest Management and REDD+' was established in December 2011 by the Collaborative Partnership on Forests⁵ (CPF), through its Global Forest Expert Panel initiative (GFEP⁶). Like previous GFEP Expert Panels, the aim of this Panel is to provide policy-relevant scientific information to intergovernmental processes and institutions related to forests and trees, thereby supporting more informed decision making by policy makers, investors, donors and other stakeholders, and contributing to the achievement of international forest-related commitments and internationally-agreed development goals. The specific objectives of this assessment, as defined by the terms of reference approved by the CPF's Global Forest Expert Panel Steering Committee, were to:

- Clarify the interactions between biodiversity⁷, carbon⁸ and forest management⁹, for different types of forests¹⁰;
- In relation to these interactions, analyse the social, economic and environmental synergies and trade-offs under REDD+ implementation;

⁵ The Collaborative Partnership on Forests (CPF) is an informal, voluntary arrangement among 14 international organisations and secretariats with substantial programmes on forests (<http://www.cpfweb.org/en/>). They collaborate to streamline and align their work and to find ways of improving forest management and conservation and the production and trade of forest products. The mission of the CPF is to promote sustainable management of all types of forests and to strengthen long-term political commitment to this end.

⁶ GFEP was established in the year 2006 within the framework of the Collaborative Partnership on Forests (CPF) and is led and coordinated by the International Union of Forest Research Organizations (IUFRO). It builds on the political recognition provided by the United Nations Forum on Forests (ECOSOC Resolution 2006/49) and the Convention on Biological Diversity (CBD Decision IX/16).

⁷ As defined by the Convention on Biological Diversity

⁸ In this assessment report 'carbon' refers to the net balance of CO₂ and non-CO₂ greenhouse gas emissions and removals.

⁹ Activities aimed at conservation, sustainable management of forests, and enhancement of forest carbon stocks to meet REDD+ intentions.

¹⁰ According to FAO definitions and FAO Global Ecological Zone classification system (FAO, 2001)

- Identify governance and policy options for REDD+ activities that capture synergies between biodiversity and carbon, and avoid perverse outcomes.

The Expert Panel was comprised of 24 scientists with recognised expertise in the biophysical and social sciences. Additional criteria for selecting Panel Members included necessary regional balance, cultural diversity and gender balance. Panel Members participated in this process in their capacity as scientific experts and did not necessarily represent the views of their institutions or organisations. In addition to the Panel Members, 18 contributing authors added their expertise to the assessment.

Authors used published, peer-reviewed scientific literature, as well as other relevant and reliable sources of information. The assessment report was subject to expert peer review prior to its completion.

1.4.2 Audience and contribution of the assessment report

A number of excellent syntheses have been published that are relevant to specific environmental, socio-economic and policy aspects of REDD+ (e.g., SCBD, 2011; Angelsen et al., 2009; 2012). This GFEP assessment report makes an important contribution to advancing REDD+ by evaluating the implications of forest management interventions under REDD+ activities in a multi-dimensional and integrated fashion and by summarising the most up-to-date scientific literature on forest biodiversity, climate change and forest

management. It seeks to provide its readers with a broad science-based perspective on relationships between forest biodiversity and carbon (and other ecosystem services) and how these complex relationships may be affected by management activities implemented to achieve REDD+ objectives. Based on this knowledge, it assesses the potential synergies and trade-offs between and among environmental and socio-economic objectives, and their relationship to governance issues at multiple scales.

In addition to synthesising the existing scientific knowledge on these topics, the report identifies areas of uncertainty and/or risk, and how these might be reduced, based on an analysis of current scientific understanding. By doing so, this assessment report seeks to provide a sound scientific basis for informed decision-making by policy-makers, investors, donors and other interested stakeholders with respect to REDD+ implementation. It is to this audience, and their scientific and technical advisors, that the report is primarily addressed.

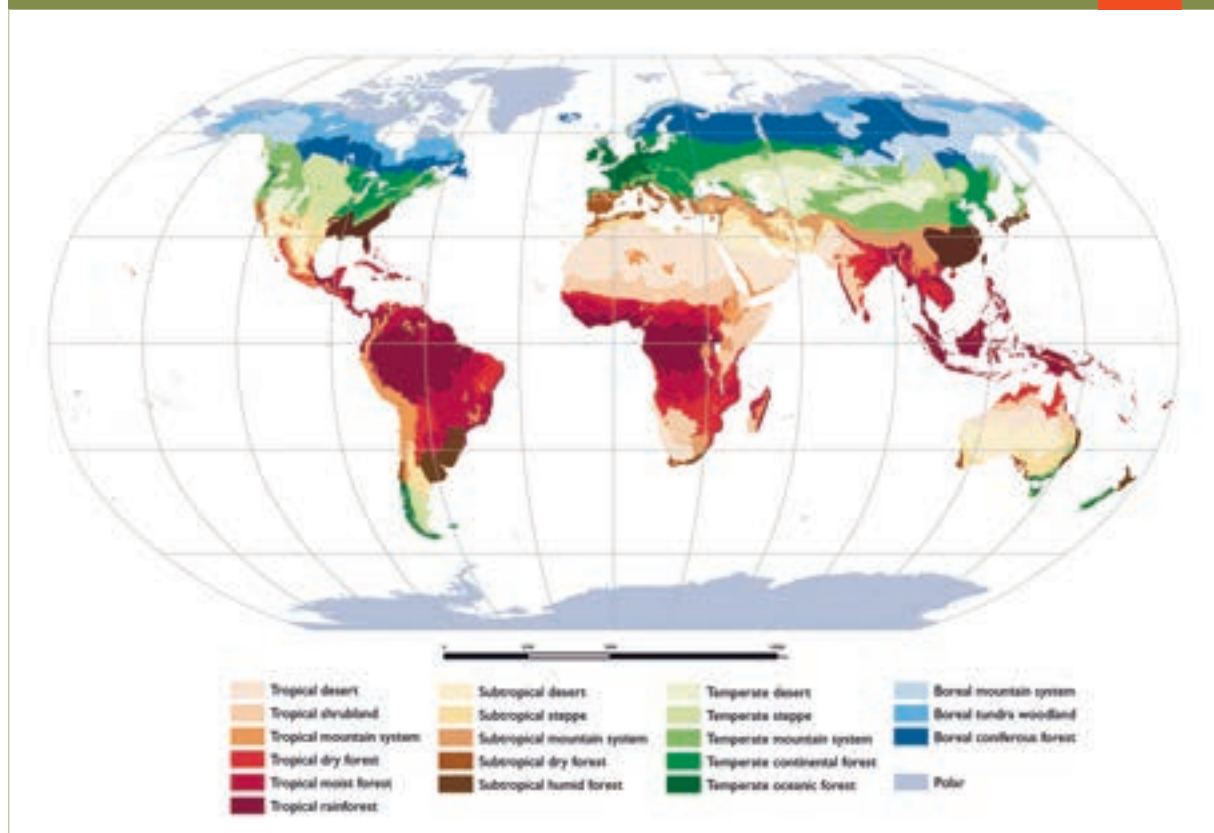
1.5 Geographical scope, scale and terminology

1.5.1 Geographical scope and forest types included in this assessment

This assessment focuses on most regions of the world in which REDD+ activities would be implemented, i.e., developing countries (non-Annex I Parties to the UNFCCC).

FAO Global Ecological Zone classification system (Iremonger and Gerrard, 2011)

Figure 1.1



Although the countries in which REDD+ activities may be undertaken include tropical, sub-tropical and temperate ecological zones, we focus on the forest types within the tropical and sub-tropical domains only, according to the FAO classification (Figure 1.1; Iremonger and Gerand, 2011). However, much of the science that underpins our understanding of forest processes, forest restoration and forest recovery applies to all forest types. The general features of the forest types in these regions are discussed in Chapter 2.

Where relevant, we also consider knowledge and experience from other regions where REDD+ activities are being planned. These would include some temperate regions, particularly those within largely (sub-)tropical countries, or the temperate, as well as montane, forest regions stretching from the Caucasus to Central Asia, the Himalayas and southwestern China.

1.5.2 Spatial and temporal scales

Existing guidance and emerging practice related to REDD+ activities do not consistently define or delimit the spatial or temporal scales over which such projects would be carried out, monitored and accounted for. This assessment report nonetheless recognises the importance of both spatial and temporal scales in its evaluations of the key questions under consideration. Throughout this report we distinguish between local (i.e., site- or stand-level) and broader (landscape-level) characteristics, relationships and impacts. For example, REDD+ activities undertaken at a management-unit level may influence carbon, biodiversity and/or non-carbon ecosystem services over a larger geographical area such as a watershed; equally, the impacts of landscape-wide management interventions may be disproportionately felt at a given site.

We also distinguish between short-term (< 20 years), medium-term (20-50 years) and long-term (> 50 years) impacts or outcomes, and their relevance for assessment of impacts of REDD+ actions on biodiversity, carbon, other ecosystem services, and environmental, economic and social synergies and trade-offs.

1.5.3 Terminology used in this report

One of the challenges related to the interpretation of UNFCCC decision language and guidance on REDD+, concerns the lack of clear, commonly-accepted definitions of some key terms. Some terms, including ‘forest’, ‘forest degradation’ and ‘sustainable forest management’ have been under discussion in international forums for many years without any broad consensus as yet regarding their definition. Key terms and phrases in the Cancún decision on REDD+ remain subject to continued debate as to their meaning (*deforestation, forest degradation*) and scope (sustainable management of forests; enhancement of forest carbon stocks).

Given this situation, the authors of this report have used definitions for key terms that, while not universally accepted, are widely recognised and used internationally, particularly within the UNFCCC, CBD, FAO and/or the

United Nations Forum on Forests (UNFF). ‘Carbon’, except where used more explicitly to refer to specific stocks and fluxes associated with forest ecosystems, refers to the net balance of CO₂ and non-CO₂ greenhouse gas emissions and removals. Biodiversity (*biological diversity*) is defined by the Convention on Biological Diversity (Article 2) as “*the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*”. In this assessment report, the focus is largely confined to forest biodiversity between species and of ecosystems (with little consideration of diversity within species). Forest management refers to the processes of planning and implementing practices for the stewardship and use of forests and other wooded land aimed at achieving specific environmental, economic, social and/or cultural objectives. Other forms of land management, including agricultural practices and land use planning that are likely to be important in REDD+ implementation, are also considered in this assessment report.

The definitions of forest, forest cover and related terminology generally follow those used by FAO’s Global Forest Resources Assessment (FAO, 2010). Terminology related to carbon stocks and fluxes generally follows that of the Intergovernmental Panel on Climate Change, except where noted. A complete listing of technical terminology used may be found in the Glossary (Appendix 2). Readers are encouraged to refer to this glossary as they read the text.

1.6 Overview of the assessment report

The structure of the report was conceptualised as a progression of building blocks which start with the ecological fundamentals of forests as they relate to biodiversity, carbon sequestration and other ecosystem services. The report then explores the different forest management options under REDD+ and seeks to highlight their main biodiversity and carbon impacts. It then considers the socio-economic dimension of these forest-related interventions and finally reviews the governance underpinnings of REDD+.

Specifically, *Chapter 2* examines the role of biodiversity in the provision of ecosystem goods and services and describes the forest types of key interest to REDD+. The chapter provides a broad overview of biodiversity and carbon relationships across the range of forest types occurring in regions where REDD+ programmes may be developed. It considers the impacts of deforestation and degradation on carbon and other ecosystem services.

Chapter 3 explores what is known about the impacts on biodiversity and carbon of the various management approaches and specific actions that are likely to be employed to achieve REDD+ objectives, based on the understanding developed in Chapter 2. The chapter identifies, insofar as possible, the circumstances under which management activities may have positive impacts on both biodiversity and carbon, and the evidence regarding

linkages, synergies and trade-offs between carbon and biodiversity objectives associated with their implementation. The chapter also examines key considerations for the design and implementation of monitoring and assessment processes, including selection of appropriate indicators, to measure and report on changes in both carbon and biodiversity.

Chapter 4 examines social and economic considerations related to REDD+, discussing how REDD+ strategies can be informed by previous land use and forest management interventions. It highlights the role of mediating factors such as structures of governance and the exercise of authority; the nature of rules and institutions for resource management; as well as types of tenure and property rights regimes, with a special focus on the most vulnerable groups. It reviews the social and economic impacts of current patterns of deforestation and forest degradation, and reviews the experience and socio-economic outcomes of previous agriculture and forest-based interventions. The chapter discusses the growing role of decentralisation and participatory forms of forest governance and management, and ways in which forest-sector interventions have attempted to incentivise behavioural change for stakeholders, drawing particularly on experience with *payments for ecosystem (or environmental) services* (PES) schemes, and forest certification. The chapter explores the implications of these previous interventions for strategies that seek to find synergies between reductions in greenhouse gases, improvements in biodiversity and positive social and economic outcomes, and identifies some key lessons of relevance to REDD+.

Chapter 5 examines the broad array of governance instruments of direct relevance to forests, carbon and biodiversity in the context of REDD+, and analyses how different actors, interests and ideas are shaping that landscape. At the international level, it considers how intergovernmental processes have generated few binding commitments and favoured strategies that enhance sovereign authority, while non-state actors have spearheaded market-based mechanisms and pressured financial institutions to develop environmental and social safeguards. This is followed by a review of international policy options and an assessment as to how these might foster synergies between REDD+ and biodiversity protection. The chapter concludes with an examination of the intersection of international forest governance with national and local agendas, and conflicting pressures for international standardisation, sovereignty and local autonomy, illustrated by case studies from Brazil, the Congo Basin, Indonesia and Nepal.

To conclude, *Chapter 6* provides a synthesis of the main findings of the assessment, and identifies key areas requiring further research.

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Chapter 2

Forest biodiversity, carbon and other ecosystem services: relationships and impacts of deforestation and forest degradation

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Abstract: REDD+ actions should be based on the best science and on the understanding that forests can provide more than a repository for carbon but also offer a wide range of services beneficial to people. Biodiversity underpins many ecosystem services, one of which is carbon sequestration, and individual species' functional traits play an important role in determining ecological processes. Higher levels of biodiversity generally support greater levels of ecosystem service production than lower levels, and ecosystem properties, such as resilience, are important considerations when managing human-modified ecosystems.

Tropical forests have high levels of biodiversity yet have experienced severe impacts from deforestation and degradation, with consequent losses of biodiversity and ecosystem processes that support the provision of ecosystem services, including carbon storage. Tropical montane and dry forests are especially vulnerable. In (sub-)tropical forests recovering from major disturbances, both carbon and biodiversity increase, but recovery rates diminish over time, and recovery of biodiversity is typically much slower than that of carbon. However, (sub-)tropical secondary forests are recognised for their biodiversity conservation values and as important carbon sinks. In many cases, anthropogenic factors – such as land use change, introduction of species or barriers to dispersal – can lead to the creation of 'novel ecosystems' that are distinct in species composition and functioning. The implications of these novel ecosystems for conserving ecological integrity and provision of ecosystem services remains poorly understood.

2.1 Introduction

A sound understanding of how *ecosystems*¹ function and the role that *biodiversity* plays in these functions is essential for the management of forests in general, and under REDD+ specifically. This chapter lays the science foundation for the suggested approaches to *forest management* and recovery under REDD+ in Chapter 3. The ecology of forest systems as it applies to the relationship between biodiversity and *ecosystem services* is discussed, with an emphasis on species, ecosystems and carbon. The first section outlines key concepts necessary to understanding the links between biodiversity and ecosystem services, including *carbon sequestration*. This relationship is examined with respect to how carbon accumulates and is lost from terrestrial ecosystems with a focus on tropical and sub-tropical forests, where the majority of REDD+ activities will be undertaken. The main (sub-)tropical forest types are presented, including their values in terms of carbon and biodiversity. The last main section outlines the effects of deforestation and forest degradation on both carbon and biodiversity.

2.2 The relationship between biodiversity and ecosystem functioning

People often think of biodiversity as a list of species without necessarily considering the roles that species perform in ecosystems. However, in recent decades, there has been an improved understanding of important linkages between species and the way that ecosystems function (e.g., Diaz et al., 2005; Aerts and Honnay, 2011; Estes et al., 2011; Cardinale et al., 2011). A large body of research has examined whether or not ongoing biodiversity loss is affecting ecosystem functioning under what is referred to as the 'biodiversity-ecosystem functioning hypothesis' (B-EF).

A sub-set of *ecosystem 'functions'* (also called '*processes*') are ecosystem services that benefit humans (see Section 2.3), including pollination, nitrogen-fixation and carbon storage (Diaz et al., 2005; 2006). Despite considerable debate over early experimental methods (e.g., Huston, 1997) and the relevance of biodiversity experiments for the biodiversity crisis (Srivastava and Vellend, 2005), there is now consensus that ecosystem functioning increases with increasing biodiversity (Chapin et al., 2000; Hooper et al., 2005; Balvanera et al., 2006). This relationship can be obscured by strong environmental effects (Laliberté and Tylianakis, 2012) and appears to be limited via competition for resources or other mechanisms at high levels of *species richness* (e.g., in *natural forests*) and depending on the scale (e.g., Mittelbach et al., 2001).

Biodiversity promotes functioning via three main mechanisms. The first is resource (or niche) complementarity (e.g., Loreau et al., 2001), whereby different species use different resources or the same resources in different ways, resulting in reduced competition. This positive effect of biodiversity becomes stronger when multiple resources are available (Tylianakis et al., 2008) and over large spatial and temporal scales because species partition resources in space or time (Cardinale et al., 2004; Zhang et al., 2011). Complementarity depends on species performing functions in different ways, thus, the strongest increase in functioning is observed when species have different functional traits (Diaz and Cabido, 2001; Fontaine et al., 2006; Hoehn et al., 2008). Furthermore, there is evidence that turnover of species among regions (Loreau et al., 2003) and evenness in the abundance of different species (Crowder et al., 2010) also promote ecosystem functioning. The second mechanism is facilitation, whereby species provide resources or alter the environment (e.g., legumes), enabling other species to perform better (Cardinale et al., 2002; Kelly, 2006). Facilitation is often used as a silvicultural tool to grow desired shade-tolerant tree

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

species beneath faster growing pioneer tree species. The final mechanism is the ‘sampling effect’, whereby there is a higher probability that a high *productivity* species will be included in a large group of species compared to a smaller group (e.g., Cardinale et al., 2006). Thus, individual species effects differ and are highly important (e.g., Diaz and Cabido, 2001; Kelty, 2006; Diaz et al., 2007) and so the loss of key species can impede forest functioning (Baker et al., 2003; Lewis, 2009).

While recent studies show that diversity of *native species* enhanced grassland productivity more than *introduced species* diversity (Isbell and Wilsey 2011), there is growing recognition of the importance of species traits (Diaz and Cabido, 2001; Fontaine et al., 2006; Hoehn et al. 2008), rather than identities, to the provision of services, suggesting that some ‘*novel ecosystems*’ (Hobbs et al., 2006; Ewel and Putz, 2004) comprised of new species assemblages may function adequately. Therefore, the functional argument for biodiversity conservation does not necessarily depend on reinstating previous ecological conditions, although provisioning, cultural, aesthetic and other benefits or services are often enhanced by native biodiversity (see Section 2.3).

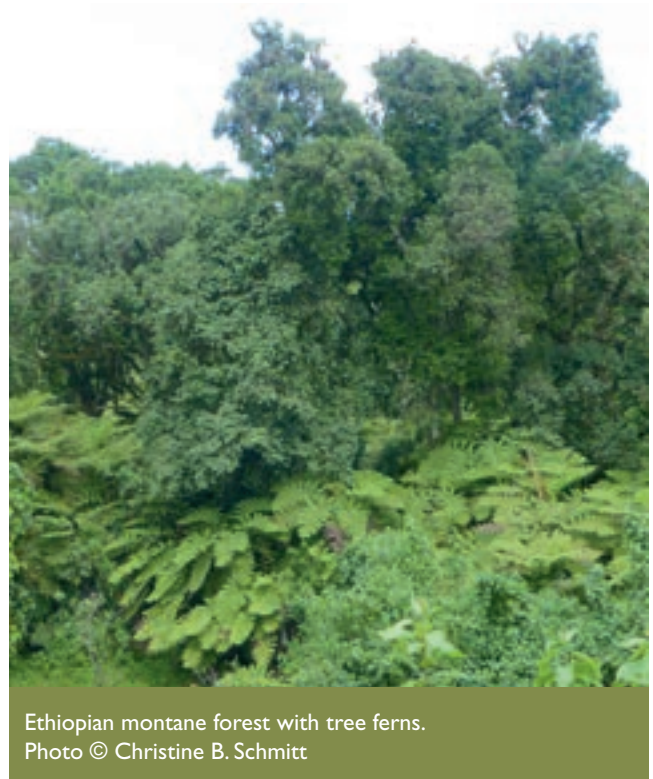
2.2.1 Biodiversity and ecosystem resistance and resilience

The ability of an ecosystem to withstand environmental change, maintain its structure and composition of species (i.e., its state), and support the provision of *services* consistently over time is referred to as ‘*ecosystem stability*’. The term ‘stability’ encompasses a suite of measures (Ives and Carpenter, 2007) including the ability of a system to remain unchanged in the face of chronic perturbations (i.e., ‘*resistance*’) and its ability to return to its original state after being altered (i.e., ‘*resilience*’) (Ives and Carpenter, 2007), although with considerable variation in rates of processes over time. In forests, stability varies among types and especially over space, but usually refers to the recognisable mix of dominant tree species (e.g., Drever et al., 2006; Thompson et al., 2009). Ecosystem stability enables some prediction of responses to management but also suggests that *ecological thresholds* exist, beyond which the system may become unstable and shift to alternate stable states (Andren, 1994; Scheffer et al., 2001; Groffman et al., 2006), with unpredictable outcomes that may produce different or reduced services (e.g., Grau et al., 2003; Chazdon, 2003; Lewis, 2009).

Functional *redundancy* among species or genotypes can help to buffer the impacts of environmental changes (Walker, 1992; Lavorel, 1999; Yachi and Loreau, 1999; Hughes and Stachowicz, 2004), and thereby help to maintain ecosystem functioning in the face of disturbance (e.g., Elmqvist et al., 2003). This redundancy also means that some species may be lost with limited effects on functioning (Walker, 1992). The strength of the buffering capacity depends on the abilities of individual species to respond to environmental fluctuations, the specific nature of their responses and the number of species (i.e., Yachi and Loreau, 1999; Elmqvist et al., 2003; Winfree and

Kremen, 2009). Buffering capacity is further affected by the condition of the ecosystem (Thompson et al., 2009): degraded systems often have reduced species richness and can have lower resilience than systems with greater integrity. On the other hand, degraded *forest ecosystems* are also often highly stable. For example, degraded systems dominated by *invasive alien species* (i.e. *Acacia* spp.) in South Africa produce greatly reduced goods and services compared to natural forests, but are highly stable and very resistant to change (van Wilgen et al., 2001).

Resistance refers to the capacity of the system to maintain its state under chronic small-scale perturbations. Some studies have suggested limited or no relationship between resistance and *species diversity* (e.g., DeClerck et al., 2006), others have suggested a positive effect (Proença et al., 2010; Royer-Tardif et al., 2010). Differences in population responses across species may produce an averaging effect that stabilises overall community functioning (Yachi and Loreau, 1999). Hence any effects of increasing biodiversity on resistance may be ecosystem-dependent and are uncertain.



Ethiopian montane forest with tree ferns.
Photo © Christine B. Schmitt

There is, however, a positive relationship between diversity (genetic, species and landscape-level) and ecosystem resilience (i.e., recovery after a major disturbance) (Pfisterer and Schmid, 2002; Griffen et al., 2009; Thompson et al., 2009; van Ruijven and Berendse, 2010). It is likely that functional diversity, not total species richness, is most relevant to ecosystem resilience (Diaz and Cabido, 2001; Fitter et al., 2005; Laliberté et al., 2010). Forest resilience is of particular interest owing to current *climate change* effects on forest processes and on carbon storage, and the resulting feedbacks to climate change. More

diverse forests are generally more resilient than forests with lower diversity, on similar sites (reviewed in Stone et al., 1996; Thompson et al., 2009). This resilience is, in part, because interactions within communities play a key role in determining the stability of the ecosystem as a whole (e.g., Balvanera et al., 2006), such as via redundancy in food web interactions (Laliberté and Tylianakis, 2010). Catastrophic impacts on ecosystems following large disturbances can be mitigated by ensuring diversity at landscape scales, since different stand types will exhibit different levels of *vulnerability* (e.g., Gunderson and Holling, 2002; Peterson, 2002). These findings suggest that the structure of entire landscapes should be considered for ecosystem management in order to maximise spatial and temporal insurance (Loreau et al., 2003; Tschardt et al., 2005). Finally, genetic diversity can also provide a considerable contribution to ecosystem resilience (Gregorius, 1996; Hughes and Stachowicz, 2004; Reusch et al., 2005). Thus, resilience is an emergent property of forest ecosystems conferred at multiple scales, through genetic, species and landscape heterogeneity (Thompson et al., 2009).

2.2.2 Ecological thresholds and safe operating space for management

Environmental change and human activities that cause local extinctions of species and alter key ecological processes may destabilise a forest ecosystem (e.g., Folke et al., 2004; Ims et al., 2007). For example, loss of species in systems can have large consequences that result in *trophic cascades*, significantly altering ecosystem structure and function (e.g., Morris et al., 2005; Estes et al., 2011). Often, ecosystem responses to environmental change may be undetectable until an *ecological threshold* is passed, resulting in non-linear and unexpected changes that may be irreversible (Andren, 1994; Scheffer and Carpenter, 2003; Folke et al., 2004; Pardini et al., 2010). Over long enough time periods or under human manipulation, ecosystems move to alternate stable states that reflect new environmental conditions (e.g., Gunderson, 2000) and may be difficult to recover (e.g., van Wilgen et al., 2001; Chazdon, 2003; Fukami and Lee, 2006), as will undoubtedly be the case under current climate change (e.g., Fischlin et al., 2009). Managing ecosystems within a ‘safe operating space’ ensures that they do not reach such irreversible levels of change. There are many examples of forest recovery to new states following *degradation* and these ‘novel’ systems may or may not provide the same ecosystem goods and services as past forests (e.g., Richardson, 1998; van Wilgen et al., 2001; Chazdon, 2003; Grau et al., 2003; Lewis, 2009).

2.2.3 The relationship between forest area and biodiversity

Several models and theories have helped to improve our understanding of the relationship between biodiversity and land use change (MacArthur and Wilson, 1967; Hanski, 1998; Ricketts, 2001). In particular, the ‘power model’ (Arrhenius, 1920), one of several alternative models used

to describe the relationships between species and area (Tjørve, 2003), has been widely applied to predict biodiversity losses driven by deforestation (Brooks and Balmford, 1996; Brooks et al., 2002; Brooks et al., 2003). The ‘species-area model’ has undergone some recent improvements to better reflect real land use changes. In particular, the differential responses of species to the landscape matrix (i.e., land uses that have replaced original forests), the effects of *forest fragmentation* and *edge effects* can now be modelled and predicted (e.g., Koh and Ghazoul, 2010; Koh et al., 2010). These theoretical considerations formalise the almost ubiquitous observation that large contiguous forest areas contain more biodiversity (especially species) than smaller and isolated stands. This pattern, coupled with current knowledge on the relationships between biodiversity and the provision of ecosystem goods and services (Section 2.3), including carbon storage and sequestration (Section 2.3.1.), reinforces the value of conserving or restoring large areas of forest to improve mitigation of forest biodiversity loss, and conservation and enhancement of *carbon stocks* (see Chapter 3).

2.3 The relationship between biodiversity and ecosystem goods and services

There are four broad categories of ecosystem services: provisioning, such as production of fibre, food and water; regulating, such as climate regulation, erosion control and pollination; supporting, such as nutrient cycling, soil formation and primary productivity; and cultural, such as spiritual and recreation benefits (MA, 2005; Diaz et al., 2005). Biodiversity is related to the provision of many of these services (Turner et al., 2007). Several studies have established explicit links between biodiversity and: pollination (Fontaine et al., 2006; Hoehn et al., 2008; Tylianakis et al., 2008), predation (Ives et al., 2005; Snyder et al., 2006; Tylianakis et al., 2008), decomposition and other soil processes (Naeem et al., 1994; Culman et al., 2010; Laliberté and Tylianakis, 2012), and *biomass* production in forests (e.g., Wardle et al., 2012). The economic value of these services has been quantified in some cases (Thompson et al., 2011), for example, the global biological control of crop pests by natural enemies is estimated to be worth USD 4.5 billion per year (Losey and Vaughan, 2006). Other services, however, such as regulation of erosion and water purification are only weakly related, or unrelated, to species diversity (MA, 2005; Dobson et al., 2006) but rather depend on the type of ecosystem and its condition (Table 2.1).

A general characteristic of ecosystem services that are strongly related to biodiversity is that the key processes occur at local scales (e.g., pollination, biological control of pests, soil formation), whereas ecosystem services and goods to which biodiversity contributes less (e.g., water quality, erosion control, oxygen production) tend to operate at larger landscape to regional scales (MA, 2005; Kremen, 2005; Maass et al. 2005; Guariguata and Balvanera, 2009).

Integrated, multiple use landscape management, (see Section 2.5.5), can maintain much of the local forest

Species richness and biodiversity relationship to ecosystem services		
Ecosystem service	Mechanism/management effects on service	Relationship w/species richness
Erosion control	Coverage of soil surface; soil retention on slopes (Pimentel et al. 1995)	None to low
Nutrient cycle	Photosynthesis, nitrogen fixation, food-web, decomposition (CO ₂ is not included here - Vitousek and Sanford, 1986; Bonan and Shugart, 1989)	medium to high (Giller, 1995; Pimentel et al., 1995; Foley et al., 2005 ; Diaz and Cabido, 2001; Eisenhauser et al., 2011; Wardle et al., 1997 ; Hättenschiler, 2005)
Natural hazard prevention: flooding	Interception of rainfall and evaporation of water infiltration by soil (FAO and CIFOR, 2005; Guillemette et al., 2005; Bruijnzeel, 2004)	None to low (Bruijnzeel, 2004)
Air quality regulation	Air filtration by plants (Givoni, 1991; Weathers et al., 2001; Bolund and Hunhammar, 1999)	Low (Givoni, 1991; Bolund and Hunhammar, 1999)
Climate regulation	Regulation of moisture in air; prevention of greenhouse gas emission (e.g. Houghton et al., 2001; Bolund and Hunhammar, 1999)	Low (Ellison et al., 2005)
Water purification and fresh water supply	Purification from polluted/contaminated to fresh water (Neary et al., 2009; Foley et al. 2005; Postel and Thompson, 2005)	Low
Disease regulation	Vector regulation, relative (lower) density of host in ecosystem/ community to regulate density of pathogens (LoGiudice et al., 2003)	High (LoGiudice et al., 2003)
Cultural services including cultural diversity and identity, recreation and ecotourism, and education	Provisioning of landscape (scenery); Symbolic (flagship) species	High but different locally
Food, fibre, timber production	Harvest and cultivation	Low to high (Thompson et al., 2009; Cardinale et al., 2011)
Pollination	Pollen transfer by animals (insects, birds) (e.g. Ricketts, 2004); forest habitat required for pollinators and depends on movement capability and landscape pool of pollinators (Kremen et al., 2004; Tschamtkke et al., 2005; Tylianakis et al., 2008)	High (Kremen et al., 2002; Ricketts, 2004; Greenleaf and Kremen, 2006; Hoehn et al., 2008; Tylianakis et al., 2008)
Biological pest control	Requires habitat for natural enemies (Landis et al., 2000), predator diversity can depend on the environmental context (Terborgh et al., 2001; Tylianakis et al., 2008; Tylianakis and Romo, 2010)	High (Ives et al., 2005; Snyder et al., 2006 ; Tylianakis et al., 2008; Tylianakis and Romo, 2010)
Seed dispersal	Fruit feeding and dispersal of seeds, usually by birds or mammals (Tschamtkke et al., 2008); diversity of dispersers can improve provision of this service (Garcia and Martinez, 2012).	None (wind) to high (animals) (Garcia and Martinez, 2012)

Table 2.1

biodiversity and provide the agents necessary for certain ecosystem services, such as pest control and pollination, thereby resulting in both sustainable agriculture and forestry (Tscharntke et al., 2005; Kleijn et al., 2006; Koyanagi et al., 2009). In contrast, large-scale intensive land conversion for timber, pulpwood or agricultural crops can degrade natural ecosystems (Matson et al., 1997; Lambin et al., 2001; Tilman et al., 2002).

2.3.1 Biodiversity and carbon sequestration and storage in forests

The sequestration and storage of carbon is one of the many ecosystem services supported by biodiversity. Carbon is initially sequestered through photosynthesis before being transferred to one of a number of terrestrial pools including *above-ground biomass* (AGB), *dead wood*, litter, roots (*below-ground biomass*) and soil (Figure 2.1). These pools are then subject to gains and losses depending on rates of growth, mortality and decomposition that are in turn affected by varying human and natural disturbances. Species can affect the long-term balance of carbon gains and losses in ecosystems through different components of the *carbon cycle*, including the magnitude, turnover and longevity of carbon stocks in soils and vegetation (Díaz et al., 2009; Murphy et al., 2008; Barantal et al., 2011; Maestre et al., 2012). Experiments with tree *plantations* of native and introduced species have often found significant and positive effects of species richness on different components of the carbon cycle, including productivity (e.g., Piotta, 2008; Healy et al., 2008), decomposition (Scherer-Lorenzen et al., 2007), soil respiration (Murphy et al., 2008) and plant mortality (Healy et al., 2008). Studies have shown that, under similar conditions, tree plantations with two or more species may achieve higher levels of productivity than single-species plantations for a range of species combinations, if species' mixes involve

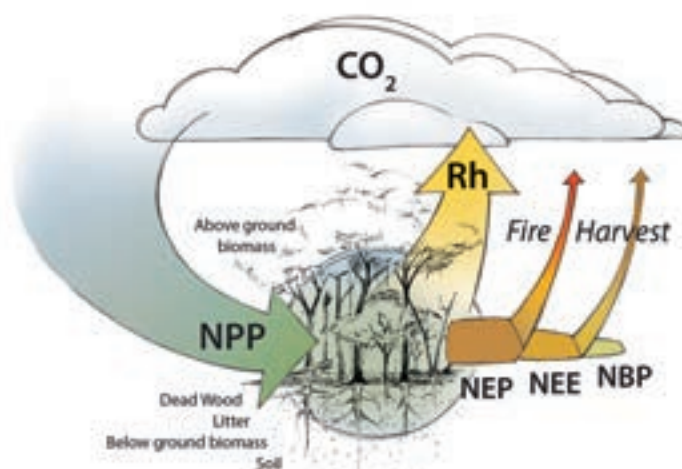
complementary resource use (i.e., stratified canopy structures) and/or facilitation of tree growth of one species by the other (Piotta, 2008; Forrester et al., 2006; Kelty, 2006) (see also Chapter 3, Section 3.2.4).

While a positive relationship between tree species richness and above-ground productivity has often been found (Thompson et al., 2009; Nadrowski et al., 2010; Potvin et al., 2011, Annex 2.1), this relationship is not universal (Mittelbach et al., 2001; Thompson et al., 2009). For example, no relationship was found between species richness and either above-ground or below-ground biomass or litter in naturally regenerating conifer stands in China, ranging from 5 to 310 years (Zhang et al., 2011), and no significant relationship was found between productivity and diversity in plantations in Australia where stands varied from 23 to 72 years (Firn et al., 2007). Nevertheless, a majority of the studies assembled in Annex 2.1 suggested a positive relationship between species richness and some aspect of forest *production* or respiration, and Zhang et al. (2012) found a positive relationship in multiple studies via meta-analysis. There are also non-linear landscape-level effects because changes in ecosystem processes are affected by the fragmentation of forests and edge effects that exacerbate species loss, population decline and ecosystem functioning (e.g., Laurance et al., 2007).

The majority of studies have not distinguished the effects of species composition on productivity and other ecosystem functions, from the effects of species richness or individual species. However, a significant positive effect of species composition, but not of species richness, was found for litter decomposition rates in rainforests (Giesselmann et al., 2010; Barantal et al., 2011). By contrast, a greater effect of species richness than species composition was reported in a natural tropical forest in Panama (Ruiz-Jaen and Potvin, 2010). The latter study reported that differences in plant species richness were more important in explaining patterns of carbon storage

The major carbon fluxes in forest ecosystems

Figure 2.1



Net Primary Production (NPP) quantifies the amount of organic matter produced annually. Most of this carbon uptake is offset through losses from the decomposition of litter, dead wood and soil C pools (R_h = heterotrophic respiration). The net balance (Net Ecosystem Production, NEP) is further reduced through direct fire emissions to yield Net Ecosystem Exchange (NEE), from which harvest losses are subtracted to estimate the annual C stock change in forest ecosystems (Net Biome Production, NBP). Positive NBP indicates increasing forest carbon stocks, a sink from the atmosphere, while negative NBP indicates a carbon source. NEE is reported from the perspective of the atmosphere and has the opposite sign convention.

Figure provided by Avril Goodall, CFS.

than changes in plant species composition or relative species dominance (Annex 2.1). In relatively simple forest systems, individual species may dominate processes, and in complex systems, certain species and *functional groups* are often particularly important in controlling specific processes (e.g., Baker et al., 2003; Diaz et al., 2007; Aerts and Honnay, 2011). Greater clarification of the importance of individual species effects, and the role of functional groups for carbon storage is an important area for further research.

2.4 Biodiversity and carbon in major (sub-)tropical forest types

2.4.1 Definition and distribution of (sub-)tropical forest types

A number of forest types can be described in the tropical and sub-tropical regions where REDD+ activities could take place (see map in Chapter 1, Figure 1.1). According to FAO definitions (2001), the tropical domain is located between the Tropics of Cancer and Capricorn (23°N to 23°S) with mean annual temperatures above 18°C. The sub-tropical domain is located between 25° and 40° north and south of the equator and the temperature is above 10°C for at least eight months of the year. The domains are further divided into ecological zones mainly based on climatic factors (Figure 1.1; FAO, 2001). For simplification and for the purposes of this assessment report, the different tropical and sub-tropical ecological zones that contain forests were combined into three major forest types: tropical rainforests, (sub-)tropical dry forests and woody savannahs, and (sub-)tropical montane forests (Table 2.2). Mangroves and freshwater swamp and peat forests are discussed separately because of their particular importance for carbon storage. The latter occur in several (sub-)tropical ecological zones. While there are estimates of the global extent of mangrove forests, the area of freshwater swamp and peat forests is uncertain (Page et al., 2010; Table 2.2). Among the different (sub-)tropical forest types, rainforests cover the largest area (at a 25 percent tree cover threshold), including primary and *secondary forests* (see Annex 2.2).

Different definitions and measurements of (sub-)tropical forest area and types render a detailed comparison across studies difficult (e.g., Mayaux et al., 2005; Schmitt et al., 2009; Mace et al., 2005). Crucial methodological differences are related to the identification of woody land cover other than natural forest and the use of different tree cover thresholds (between 10 and 40 percent) that influence the estimation of extent, especially for (sub-)tropical dry forests and savannahs (Schmitt et al., 2009; Miles et al., 2006). For instance, the tropical dry forest area of 707 Mha estimated by Mayaux et al. (2005) is much larger than the 458 Mha identified in Table 2.2. Furthermore there are many other different global ecosystem classifications, such as the Global Land Cover 2000 classes (see Mayaux et al., 2005) and the WWF ecoregions, based on bio-geography and species assemblages (Olson et al.,

2001; Mace et al., 2005). There are also many finer-scaled forest classifications, using plant species composition and environmental factors that were developed for the sub-national (e.g., Clark and Clark, 2000; Cannon et al., 2007) or national level (e.g., Friis et al. 2010; Letouzey, 1985).

2.4.2 Spatial patterns of biodiversity in (sub-)tropical forest types

There is broad consensus that species richness is generally highest in tropical rainforests compared to all other (sub-)tropical forest types (Table 2.3; Mace et al., 2005). However, species richness is only one aspect of biodiversity, and it is crucial to consider species composition, species distributions and the differences in species composition across similar forest types but in different regions of the world. For example, there are notable differences in the vascular plant and vertebrate species richness among the tropical rainforests of Africa, Asia and South America. In addition, there are areas of extremely high vascular plant and vertebrate species richness in tropical montane forests and the number of tree species is higher in (sub-)tropical moist montane forests than in (sub-)tropical moist lowland forests (Table 2.3).

Endemism is very high in (sub-)tropical forests but patterns of species richness and endemism are not congruent among all continents or major forest types (Gentry, 1992; Orme et al., 2005; Ghazoul and Sheil, 2010). While high diversity tropical rainforests are concentrated in lowland areas, with high and evenly distributed rainfall, the highest rates of endemism occur in isolated cloud forests, topographically dissected montane areas and on islands or other isolated forest areas (Gentry, 1992). Perhaps the best available data are for birds, which indicate that 32, 24 and 15 percent of global endemic avian species occur in tropical lowland moist, tropical montane moist and tropical dry forests, respectively (Stattersfield et al., 1998). Many (sub-)tropical forest areas are recognised as global biodiversity ‘hotspots’ because they feature exceptional concentrations of endemic species and are experiencing exceptional loss of *habitat* (Myers et al., 2000; Mittermeier et al., 2004; Schmitt et al., 2009). For example, Hubbell et al. (2008) suggested that there are over 11,000 tree species in the Amazon region, but at current rates of deforestation, forest degradation and climate change, at least 1,800 to 2,600 species are predicted to become extinct in the next few decades. In fact, habitat change and loss are the major reason for all groups of species to be listed as vulnerable and endangered on the IUCN Red List of Threatened Species (Vié et al., 2009). (Sub-)tropical moist montane, (sub-)tropical moist lowland and (sub-)tropical dry forests contain the greatest percentage of species affected for all taxa (32, 22, and 22 percent, respectively) (Table 2.3).

Tropical rainforests

The global distribution of tropical rainforests is primarily determined by climatic conditions such as uniformly high temperatures, high precipitation of at least 1,500 mm yr⁻¹ (but mostly between 2,000 and 3,000 mm yr⁻¹) and a

Table
2.2

Forest area and above-ground biomass (AGB) carbon for the five major forest types in the tropical and sub-tropical domain based on the Saatchi et al. (2011) tropical above-ground biomass map, the FAO ecological zones (Figure 1.1; FAO, 2001) and the MODIS forest cover map (25 percent forest cover threshold) (NASA, 2010). Carbon is defined as 50 percent above-ground biomass. SD is the standard deviation of spatial variations of estimates across the regions. For area and carbon data by region and ecological zone see Annex 2.2.

	Tropical rainforests	(Sub-) tropical dry forests and woody savannahs	(Sub-) tropical montane forests ¹	(Sub-) tropical Freshwater swamp and peat forests ²	(Sub-) tropical mangrove forests ²
Combined FAO ecological zones	Tropical rainforest	Tropical moist deciduous forest, Tropical dry forest, Tropical shrubland, Sub-tropical dry forest	Tropical mountain systems, Sub-tropical mountain systems		
Forest area across Africa, Latin America and Southeast Asia (Mha)	1,101.6	457.9	164.2	51.9 (tropical flooded forests, Mayaux et al., 2005) ³	15.2 (global mangrove forests, FAO, 2007; Spalding et al, 2010)
AGB carbon density (Mg C ha ⁻¹ ± SD)	115 ± 79	53 ± 121	94 ± 87	not known (NB: high below-ground carbon density)	not known (NB: high below-ground carbon density)
Total AGB carbon across Africa, Latin America and Southeast Asia (10 ⁶ Mg)	126.1	24.4	15.5	not known	not known
Examples	Amazon Basin, Congo Basin	Cerrado (South America), Miombo (Africa)	Eastern African mountains, Eastern Himalayas	Brazil, Borneo, Sumatra	Gulf of Guinea (Africa), Greater Sundas (Asia)

NB: The sub-tropical humid forests are not included in this summary table because they encompass a wide range of forest ecosystems from evergreen broadleaved forest in Southeast China to coniferous forest in Brazil and bushland in southern Africa. For area and carbon data see Annex 2.2.

¹ Mountain systems are located at > 1,000-1,500m elevation (FAO, 2001)

² These are azonal forest types that can occur within the other FAO ecological zones and are not mapped out explicitly (Figure 1.1; FAO, 2001). Area data are from other sources as indicated.

³ Area of tropical forest regularly flooded by freshwater and saline water (10 percent tree cover threshold); includes mangrove forests but peat forests probably underestimated (e.g. Page et al., 2010).

short or absent dry season (see map in Chapter 1, Figure 1.1; FAO, 2001). Competition for light is the primary driver of vegetation dynamics and structural complexity (Murphy and Bowman, 2012). The high species diversity of tropical rainforests renders small-scaled classification of forest types complex (Leigh et al., 2004; Ghazoul and Sheil, 2010).

There are notable differences in rainforest species diversity among the continents. For example, there is high bird and bat species richness and many Bromeliads mostly in the Neotropics, while most diversity of gliding animals and dipterocarp trees occurs in Southeast Asia. The estimated number of rainforest plant species also varies, with 93,500, 61,700, and 20,000 species in the Neotropics, the Asia-Pacific region and Africa (including Madagascar),

respectively (Corlett and Primack, 2011). These differences are related to continental drift, differences in rainfall and its seasonal distribution, and extinctions caused by past natural and anthropogenic environmental changes (Corlett and Primack, 2011; de Gouvenain and Silander, 2003; Parmentier et al., 2007). Within the tropical rainforest regions, lower species diversity occurs where there is annual rainfall under 2,000 mm yr⁻¹, a pronounced dry season, periodic flooding, and sandy or peat soils (FAO, 2001; Corlett and Primack, 2011).

(Sub-)tropical dry forest and woody savannahs

(Sub-)tropical dry forest and woody savannahs contain several ecological zones (Table 2.2), characterised by a distinct dry season of at least three, but up to eight,

Numbers of species by selected species groups and major (sub-) tropical forest types from IUCN (www.redlist.org) that have been assessed, and total species numbers in vulnerable (=VU) or higher categories (critically-endangered and endangered = EN).

Table
2.3

Major forest type	Mammals		Birds		Amphibians		Reptiles		Trees	
	all sp.	EN/VU	all sp.	EN/VU	all sp.	EN/VU	all sp.	EN/VU	all sp.	EN/VU
(Sub-) tropical dry	1014	179	1853	161	422	93	642	137	694	452
(Sub-) tropical moist lowland	2259	511	5045	557	2946	724	934	191	1131	732
(Sub-) tropical mangrove	172	61	912	62	15	2	47	9	10	4
(Sub-) tropical swamp	268	70	688	44	132	15	27	3	65	14
(Sub-) tropical moist montane	1551	417	3627	450	2831	1197	627	129	1479	1085

NB: There are some differences between the way forests are classified under the IUCN Red List and the FAO global ecological zones (Figure 1.1). For example, Brazilian cerrado and African miombo woodlands are classed mostly as savannah ecosystems under the IUCN, while FAO (2001) includes these woodland types with tropical moist deciduous and tropical dry forests. Thus, the (sub-) tropical dry forests and woody savannahs (Table 2.2) are actually more species rich than the IUCN (sub-) tropical dry forest in this Table would suggest.

months and annual rainfall mostly below 1,500 mm yr⁻¹ (FAO, 2001). In addition to climate, the distribution of these forest types is governed by soil fertility and fire frequency (Murphy and Bowman, 2012). There are areas of dense (sub-)tropical dry broadleaf forest where soils are relatively fertile but where long dry periods, or decadal scale droughts, occur, e.g., the monsoon forests of mainland Southeast Asia, the Atlantic dry forests of Brazil and the coastal forests of Southern and Eastern Africa (Olson

et al., 2001; Burgess and Clarke, 2000; FAO, 2001). These forests are often species rich with many endemics; in Mexico, the dry broadleaf forests contain about 6,000 vascular plant species, of which 40 percent are endemic (FAO, 2001).

Where soils are less fertile or extended dry periods more frequent, (sub-)tropical dry forests can structurally resemble woody savannahs, but lack a significant grass component (Grace et al., 2006; Vieira and Scariot, 2006; Pennington et al., 2009). Areas of varying forest cover are often closely interconnected with savannahs, whose distribution is determined by a poorly understood combination of nutrient-poor soils, natural or anthropogenic fires, and wild or domestic animal grazing (Murphy and Lugo, 1986; Murphy and Bowman, 2012; Prance, 2006). An example is the Brazilian *cerrado* (a mosaic of grasslands, savannah, woodlands and patches of gallery forest), where growth of closed dry forest is inhibited by low soil fertility, despite relatively high annual rainfall (Grace et al., 2006). In Central and Southern Africa, the structurally similar woody savannahs are characterised by the *Brachystegia* (miombo woodlands) (Prance, 2006; Shirima et al., 2011). Both the *cerrado* and the miombo woodlands harbour large numbers of endemic species (Mittermeier et al., 2003; 2004).

(Sub-)tropical montane forests

(Sub-)tropical montane forests are located between 1,000 and about 4,000 m, and support different forest ecosystems along altitudinal belts from evergreen sub-montane rainforest to cloud forest (Figure 1.1; FAO, 2001). Generally, forest canopy height declines with increasing altitude; the (sub-)tropical timberline (where shrubs and grasslands dominate) depends on climate and anthropogenic influence but is located at 3,000-4,000 m (e.g., Bussmann, 2004; Friis et al., 2010; Kessler, 2000).



Madagascar montane forest
Photo © Christine B. Schmitt

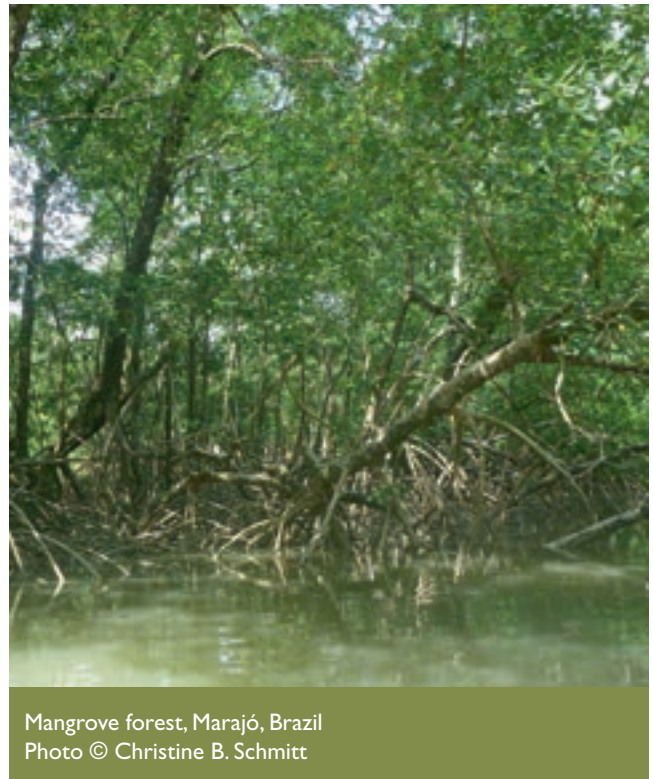
Regions of high spatial heterogeneity, such as the mountainous areas of the humid tropics and sub-tropics support high biodiversity, including many endemic species (Mace et al., 2005; Gentry, 1992). For instance, the tropical broadleaf forests in the mountains of Costa Rica have an estimated 10,000 vascular plant species per 10,000 km² (Olson et al., 2001). Conifer montane forests occur in Mexico and Central America, the Himalayas and the dry forests of Ethiopia (FAO, 2001). Many of the (sub-)tropical mountain areas are designated as biodiversity hotspots (Myers et al., 2000; Mittermeier et al., 2004) including in the tropical Andes and the montane regions of Eastern Africa (Schmitt et al., 2009).

(Sub-)tropical freshwater swamp and peat forests

Freshwater swamp forests are characterised by waterlogged soils (FAO, 2001). Globally, they are most extensive in the Amazon basin where two types occur: *igapo* that is more or less permanently flooded swamp forest, and *várzea* that is seasonally flooded with relatively rich soils (Butler, 2011). Other areas occur in the Congo River basin, New Guinea and Southeast Asia. Peat and swamp forests generally support a lower diversity of plants than other rainforests on drained land because plants need special adaptation mechanisms for seasonal water extremes (Corlett and Primack, 2011; FAO, 2001). Peat forests occur on raised, deep peat beds where the forest is isolated from ground water and are particularly extensive on the islands of Borneo, Sumatra and New Guinea. The Amazonian peatland is also significant in area (Lähteenoja et al., 2009; Page et al., 2010). Height and species diversity of the vegetation decreases with increasing peat depth, and forests can be dominated by one species, such as the *alan* peat swamp forests of Borneo dominated by light red meranti (*Shorea albidia*) (Corlett and Primack, 2011; FAO, 2001; Rieley and Page, 1997).

(Sub-)tropical mangrove forests

Mangrove forests are formed by a highly specialised group of trees and shrubs that thrive in saline, tidally flooded soils, along (sub-)tropical coasts and estuaries (Spalding et al., 2010; Giri et al., 2011) and have a simple structure and relatively low plant diversity (Corlett and Primack, 2011). At least 73 mangrove species and hybrids are recognised worldwide, with 38 common species. Highest richness of mangroves is concentrated in a core area around insular Southeast Asia (Spalding et al., 2010). Many mangrove stands are dominated by few species, with distinctive communities zoned by substrate age, salinity and tidal conditions. Faunal diversity is also relatively low but the tidal influence creates a unique combination of marine, estuarine and terrestrial species (Hogarth, 2007).



Mangrove forest, Marajó, Brazil
Photo © Christine B. Schmitt

2.4.3 Spatial patterns of carbon in (sub-)tropical forest types

On average, tropical rainforests have higher above-ground biomass carbon density than the other (sub-)tropical forest types, and globally they are the largest pool of above-ground biomass carbon (Table 2.2). The data on above-ground biomass and total biomass carbon for (sub-)tropical forests by FAO ecological zones presented here is the first developed using globally consistent methods (Annex 2.2). There is a high variability in carbon density within the major (sub-)tropical forest types (Table 2.2) but also within each ecological zone (Annex 2.2), which is likely related to the inclusion of both primary and secondary forests and to the considerable variety of forest ecosystems in each ecological zone. For example, for African montane forests, above-ground biomass carbon decreases considerably above approximately 1,600 m altitude (Baccini et al., 2008; Marshall et al., 2012).

While the importance of tropical rainforests to the global carbon pool is uncontested, a detailed comparison of global carbon data is impeded by the high variability in the consideration of different carbon pools, definitions of forest types and wall-to-wall remote sensing versus plot-based studies, which are mostly not global in scale. For instance, the inconsistent use of definitions for (sub-)tropical dry forests and woody savannahs leads to a large variation in estimates of their biomass and carbon content (Becknell et al., 2012; Grace et al., 2006; Baccini et al., 2008).

A major knowledge gap remains in understanding the magnitude and dynamics of below-ground carbon stocks and fluxes in the different forest types (Lal, 2005). Saatchi et al. (2011) estimated below-ground biomass (BGB) carbon as a fraction of that above ground (Annex 2.2), a method

prone to uncertainty, as for example in tropical savannahs, where carbon content below ground may exceed that above ground due to relatively high root biomass (Grace et al., 2006). Furthermore, tropical peatland forests have organic soils up to several metres deep and are among the largest terrestrial organic carbon reserves (on a ha⁻¹ basis) (Lähteenoja et al., 2009). (Sub-)tropical mangrove forests can have relatively low above-ground biomass carbon but very high total carbon stocks because of their high investment in below-ground biomass and high *soil carbon* (Donato et al., 2011; Komiyama et al., 2008; Lovelock, 2008).

2.4.4 Congruence between carbon density and species richness across different scales and ecological zones

At the global scale, there is a strong positive correlation between the distribution of total biomass carbon and species richness of selected vertebrates (Strassburg et al., 2010), suggesting that tropical rainforest areas can deliver multiple benefits under REDD+ for both biodiversity and carbon objectives. However, there are also extensive areas with high species richness and lower carbon density such as (sub-)tropical mountain ecosystems (Strassburg et al., 2010). In contrast, (sub-)tropical flooded and peat forests may have lower species richness and lower above-ground biomass carbon but very high below-ground carbon density (e.g., Corlett and Primack, 2011; Donato et al., 2011; Lähteenoja et al., 2009). In (sub-)tropical dry forests and woody savannahs carbon content can be highly variable above- and below-ground, with high endemism (e.g., Grace et al., 2006; Mittermeier et al., 2003; 2004).

Spatial relationships in the distribution of carbon and biodiversity have also been investigated at the national scale, illustrating a high level of correspondence between carbon stocks and mammal species richness in Tanzania (Khan, 2011; see Figure 3.3 in Chapter 3) and different taxonomic groups in South Africa (Egoh et al., 2009). Data from Mexico illustrate the spatial pattern and relationship between biomass and vertebrate species richness at a very large scale (Figure 2.2).

By contrast, at the sub-national level, carbon stocks exhibited low overlap with species richness in several South African ecosystems (forests, savannahs and grasslands) when compared to services such as water flow and soil retention at the scale of 1 km (Egoh et al., 2009). Here the spatial congruence between carbon stocks and species richness was consistently low, with values of 8, 13 and 21 percent, for mammals, birds and butterflies, respectively. Similarly, Anderson et al. (2009) and van Rensburg et al. (2002) reported that associations between carbon and biodiversity were sensitive to spatial resolution, extent and regional variation in data.

Detailed knowledge of forest types, carbon and biodiversity patterns at sub-national and national scales can help to facilitate decision-making for REDD+ investments to achieve conservation and carbon objectives (e.g., Egoh et al., 2009; Cannon et al., 2007; UNEP-WCMC et al., 2008) (see also Chapter 3).

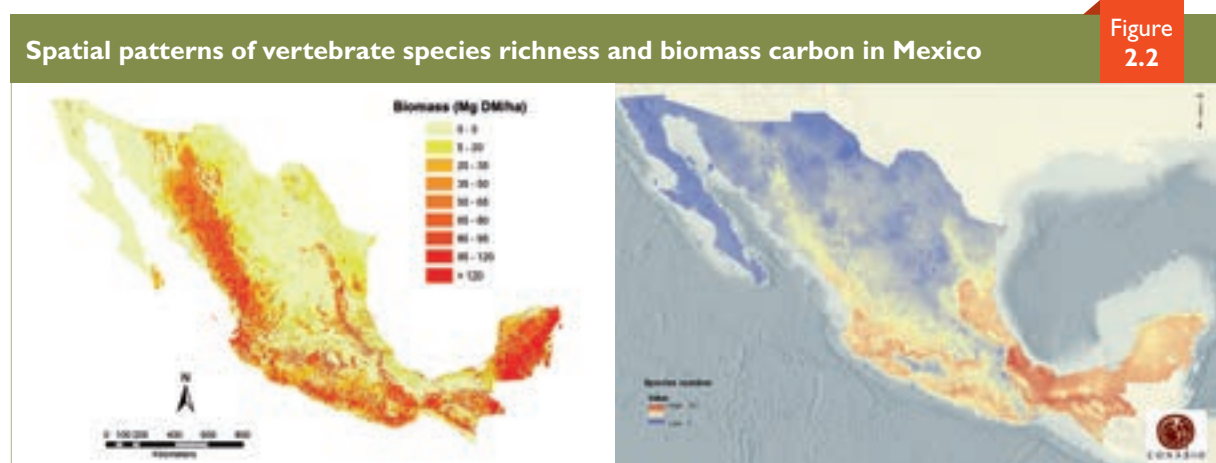
2.5 Effects of deforestation and forest degradation on carbon and biodiversity

2.5.1 Causes of global deforestation and forest degradation

Causes of deforestation

Between 1990 and 2010, 13 to 16 million ha of forests were lost each year (FAO, 2010). Rates of deforestation are particularly high in the tropical ecological domain, with an estimated net forest loss of 8.0 million ha yr⁻¹ between 2000 and 2005 (FAO, 2011). Although recent deforestation rates have fallen in some countries, continued pressure on forests would suggest that rates of forest loss in tropical and sub-tropical countries are likely to remain high in the foreseeable future (e.g., Rudel et al., 2009; FAO, 2011).

The ultimate drivers of forest loss include rapid population growth, increasing global natural resource consumption, and the often over-riding effects of economic globalisation and global land scarcity (Lambin and



The left figure shows total biomass density (dry matter in Mg ha⁻¹) distribution (Source: de Jong and Olguin, Ecosur); the right map illustrates predictive modelling of expected vertebrate species richness (Source: CONABIO).

Meyfroidt, 2011; Kissinger et al., 2012), combined with problems of weak *governance*, inadequate policies, lack of cross-sectoral coordination, perverse incentives and illegal activities (Kissinger et al., 2012 - also see Chapter 4).

Agricultural expansion has been the most important proximate cause of forest loss, accounting for 80 percent of deforestation worldwide, with the majority occurring during the 1980s and 1990s through conversion of tropical forests (Gibbs et al., 2010). Commercial agriculture has been responsible for approximately two thirds of deforestation in Latin America, while in Africa and (sub-) tropical Asia subsistence farming is the major driver of land use change (Kissinger et al., 2012). Recent human impacts on global forest ecosystems have not been equal, with some forest types now under severe threat. Both (sub-)tropical dry and montane forests have been converted to a large extent because they are located in climates highly suitable for agriculture and grazing (Rodrigues et al., 2004; Grace et al., 2006). Mangrove forests declined by 19 percent from 1980 to 2005 (FAO, 2007), due to land clearing, aquaculture, changes to hydrological regimes and coastal development (Donato et al., 2011; Spalding et al., 2010).

Causes of forest degradation

Forest degradation can be characterised as a continuum of decline in the provision of ecosystem services resulting from increasing levels of unsustainable human impacts, relative to a more desirable condition (e.g., Chazdon, 2008; Thompson et al., in press). While deforestation represents an obvious ecosystem change, forest degradation is often more difficult to discern or quantify (Sasaki and Putz, 2009). The Collaborative Partnership on Forests (CPF) broadly defines forest degradation as “a reduction of the capacity of a forest to provide goods and services” (Simula, 2009; Thompson et al., in press). Forest may be degraded from several perspectives including productive capacity, protective capacity, biodiversity, health and carbon storage, but how these perspectives on degradation are perceived is a societal decision (Thompson et al., in press). The International Tropical Timber Organization (ITTO, 2002) has estimated that up to 850 million ha of tropical forest could already be degraded.

The proximate drivers of forest degradation include unsustainable and illegal logging, over-harvest of fuelwood and *non-timber forest products* (NTFPs), over-grazing, human-induced fires (or fire suppression in dry forests) and poor management of *shifting cultivation* (Chazdon, 2008; Kissinger et al., 2012). For example, unsustainable timber extraction accounts for more than 70 percent of forest degradation in Latin America and (sub-) tropical Asia (Kissinger et al., 2012). Unsustainable logging has resulted in forests being degraded by removal of high-value trees (Putz et al., 2011), the collateral damage associated with log extraction, and subsequent burning and clearing (Asner et al., 2006; Foley et al., 2007). Fuelwood collection, charcoal production and grazing are major causes of forest degradation, particularly across Africa (Kissinger et al., 2012). For example, the miombo woodlands of Southern and Eastern Africa provide

fuelwood for approximately 100 million people (Boucher et al., 2011).

Although fire is a natural element in many forest ecosystems, humans have altered fire regimes across 60 percent of terrestrial habitats (Shlisky et al., 2009). Fires in tropical rainforests have increased in extent and frequency with the expansion of agriculture (Uhl and Buschbacher, 1985), forest fragmentation, unsustainable shifting cultivation and logging (Siegert et al., 2001; Nepstad et al., 1999; Alencar et al., 2006). Forest fires were estimated to have burned 20 million hectares of tropical forests in Southeast Asia and Latin America during 1997-1998 (Cochrane, 2003).

In addition to the detrimental impacts of land-use change and human-induced forest degradation, climate change poses an increasing threat to global forest ecosystems, in particular through an increase in the frequency of severe droughts (Malhi, 2012). (Sub-)tropical regions that appear particularly vulnerable to warming and drought include Central America, Southeastern Amazonia and West Africa (Zelazowski et al., 2011; Phillips et al., 2008).

Deforestation and forest degradation can often act synergistically. Deforestation fragments forest landscapes, which often results in degradation of remaining forests due to edge effects (e.g., drying of the forest floor, increased fire frequency, increased tree mortality and shifts in tree species composition) (Balch et al., 2008; Blate, 2005; Alencar et al., 2004; Foley et al., 2007). Poorly planned logging and deforestation increase road access to remaining forest interiors, further facilitating shifting cultivation and other land clearing, hunting, illegal logging, blowdown and fire (Foley et al., 2007; Bradshaw et al., 2009; Griscom et al., 2009; Harrison, 2011). Degradation can lead to deforestation; for example, in the Brazilian Amazon basin, Asner et al. (2006) estimated that 16 percent of unsustainably logged areas were deforested during the following year, and 32 percent in the following three years. Degraded forests can often remain in a degraded state for long periods of time if degradation drivers (e.g., fire, human and livestock pressures) remain, or if ecological thresholds have been passed, and yet remain officially defined as ‘forests’ for classification purposes (Murdiyarso et al., 2008; Sasaki and Putz, 2009; FAO, 2010).

2.5.2 Impacts of deforestation and forest degradation on carbon

Tropical and sub-tropical forests store an estimated 247 Gt C (in biomass both above ground and below ground) (Saatchi et al., 2011). When the forest is replaced by croplands, often through burning, a large portion of carbon stored in above-ground vegetation is immediately released to the atmosphere as carbon dioxide (and other *greenhouse gases*), or over time through the decomposition of debris. Carbon in soils following deforestation can also become a large source of emissions because of increased soil respiration with warmer ambient temperatures (Bormann and Likens, 1979). There is increased soil loss with higher flooding and erosion rates, with carbon being transported downstream where a large fraction of

the decayed organic matter is released as CO₂ (Richey et al., 2002).

In the last two decades, the net carbon emissions from tropical deforestation and degradation were almost equal to the total emissions from global land-use change (1.1 Pg C yr⁻¹) because effects of land-use changes on carbon were roughly balanced in non-tropical areas (Pan et al., 2011), effectively negating the role that tropical forests play as long-term *sinks* of carbon dioxide (Phillips et al., 1998; Lewis et al., 2011). Carbon emissions from forest degradation are difficult to assess because of a lack of consistent data. Forest degradation is often pooled with deforestation to estimate emissions from land-use change (e.g., Houghton, 2003), or is estimated as less than 10 percent of tropical carbon emissions (e.g., Nabuurs et al., 2007). Emissions from degradation, however, are likely to be more substantial (Putz et al., 2008; Lambin et al., 2003).

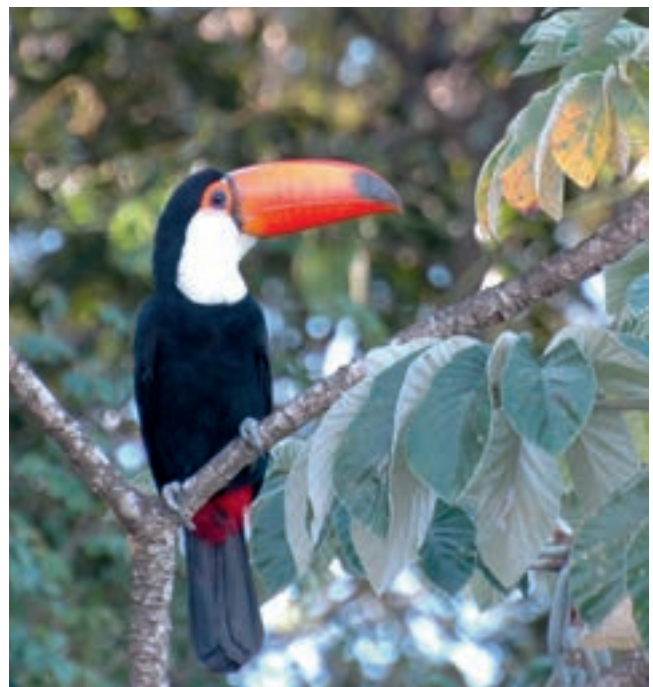
Poor logging practices create large canopy openings and cause collateral damage to remaining trees, sub-canopy vegetation and soils (Asner et al., 2006). During timber harvest, a substantial portion of biomass carbon (approximately 50 percent) can be left as logging residues, and about 20 percent of harvested wood biomass is further lost in the process of manufacturing wood products (Pan et al., 2011; Ciais et al., 2010). There is a continuing loss of carbon from oxidation of wood products, and the majority of wood products retain carbon for less than 30 years (Earles et al., 2012). Degradation of dry forests from extensive fuelwood gathering may have an impact comparable to commercial timber harvesting in rainforests (Murdiyarto et al., 2008). FAO (2006) estimated that fuelwood harvesting accounts for 40 percent of global removal of wood from forests. In recent decades, the frequency and size of forest fires have increased in many (sub-)tropical regions (Aragão and Shimabukuro, 2010), often associated with deforestation and land-use practices (Cochrane, 2003). Fire frequency may be intensified in forests that have been degraded by logging or previously burned (Holdsworth and Uhl, 1997; Cochrane, 2003), because these areas become more flammable and fire is more likely with human encroachment (Foley et al., 2007; Barlow et al., 2012).

Slash-and-burn agriculture makes a significant contribution to overall greenhouse gas emissions in tropical countries (see Chapter 3). Estimated greenhouse gas (GHG) emissions from slash-and-burn agriculture amount to 241 ±132 Tg yr⁻¹ for Asia, 205 ±139 Tg yr⁻¹ in Africa and 295 ±197 Tg yr⁻¹ in the Americas (Silva et al., 2011).

In Southeast Asia, freshwater swamp and peat forests have been severely degraded in recent decades by unsustainable logging and agricultural expansion. Significant increases in the number of large-scale forest fires have resulted in large releases of CO₂ and non-CO₂ GHG emissions in the region (Page et al., 2002; Hooijer et al., 2010), due in particular to the high carbon content in partially decayed organic matter of peat soils (Donato et al., 2011).

The consequences of fire hazard are still poorly understood but the impact on carbon emissions is particularly

significant (van der Werf et al., 2009). When measured against adjacent unburned forests, even low to medium severity fires in undisturbed or lightly degraded intact forest can kill over 50 percent of all trees (Barlow et al., 2003) and almost all of the large lianas (Cochrane and Schulze, 1999; Gerwing, 2002; Barlow et al., 2012). Trees in tropical humid forests are particularly susceptible to fire damage because fires are historically rare (Aragão and Shimabukuro, 2010; McMichael et al., 2012). In extreme drought years, carbon emissions from tropical forest fires can exceed those from deforestation (Houghton et al., 2000). For example, total estimated carbon emissions from tropical forest fires during the 1997-98 El Niño event were 0.83 to 2.8 Pg C yr⁻¹ (Alencar et al., 2006; Cochrane, 2003; Page et al., 2002).



Toucans are important seed dispersers. Toco toucan (*Ramphastos toco*) in Brazil's Atlantic Forest. Photo © PJ Stephenson.

Over decadal time scales, forests can experience a loss of carbon stocks through the indirect effects of hunting of species that have functional roles, such as pollinators (e.g., large fruit bats), seed predators (e.g., peccaries, agoutis, squirrels) and seed dispersers (e.g. primates, frugivorous bats and birds) (Brodie and Gibbs, 2009; Harrison, 2011). For example, hunting in Peru has caused a shift in tree species composition as large-seeded species (which often have a high wood density; Wright et al., 2007) dispersed by large animals are replaced by smaller-seeded species, dispersed abiotically or by smaller animals (Terborgh et al., 2010; Terborgh et al., 2008). Tree growth rates and above-ground plant productivity are positively affected by red howler monkeys (*Alouatta seniculus*) (Feeley and Terborgh, 2005). Loss of these processes results in subtle but long-term cumulative degradation of forest functioning.

2.5.3 Impacts of deforestation and forest degradation on biodiversity

Deforestation and forest degradation are the two major causes of loss of biodiversity from forests (e.g., Vié et al. 2009). Conversion of forests to permanent agriculture and pasture results in an almost total loss of the original biodiversity, with reduced ecosystem function (e.g., Gibson et al., 2011). In contrast, well-managed shifting cultivation leads to a patchy habitat mosaic of agricultural plots, fallow and forests, and has lower local carbon and biodiversity impacts than more intensive land uses (Gardner et al., 2009). As the cultivation phase is typically short (1-3 years), agricultural plots are often small (less than 1 ha) and close to either primary or older secondary forests that act as recolonisation sources, and soil compaction is limited, biomass and biodiversity can recover rapidly during the fallow phase (Gehring et al., 2005; Lamb et al., 2005).

Uncontrolled human-induced fires, such as those originating from agricultural areas or road edges, can reduce forest biodiversity, particularly in tropical rainforests. For example, large scale fires in Amazonian forests were unlikely to have occurred more than once or twice per millennium (Sanford et al., 1985; Turcq et al., 1998) and the regional flora and fauna shows little adaptation to these episodic disturbance events (Uhl and Kaufmann, 1990; Peres et al., 2003). Fires in the rainforests of Amazonia and Southeast Asia have long-term effects on the composition of the vegetation, with an increase in pioneer species and reduction or loss of mature forest species (Barlow and Peres, 2008; Cochrane and Schulze, 1999; Slik et al., 2002; 2010). A synthesis of Amazonian bird data showed that low-intensity understorey fires can alter species composition more than selective logging, causing avian species changes similar to extreme forest fragmentation (1-10 ha isolated forest fragments; Barlow et al., 2006). Fires also exacerbate the impacts of selective logging and fragmentation on biodiversity, leading to significant reductions of forest-dependent birds and large vertebrate species in fragmented landscapes (Lees and Peres, 2006; Michalski and Peres, 2005).

Unsustainable logging, especially after multiple harvests and where fire is not controlled, can precipitate a shift in forest state, including the loss of a complex canopy, domination by dense undergrowth and pioneer species, loss of important functional species and increases in the abundance of some generalist and *invasive species* (e.g., van Wilgen et al., 2001; Asner et al., 2006; Souza et al., 2005), and a generally impoverished biota (Cleary, 2003). Altered forest states may continue to provide some services but ecosystem functions of degraded secondary tropical forests and their long term successional trajectories is an area of scientific and management uncertainty (Hobbs et al., 2006).

Forests can be degraded through unsustainable hunting that results in loss of game animals for local people and loss or impairment of functional roles provided by these species (Nasi et al., 2008; Harrison, 2011; Thompson et al., in press). Such “empty forests” are common in

tropical areas, even where forests are protected (e.g., Redford, 1992; Collins et al., 2011). Vertebrate biomass can drop dramatically from around 700 kg km⁻² in non-hunted sites to 200 kg km⁻² in heavily hunted areas in the Amazon (Peres, 2000) and primate relative abundance may decline almost 10-fold in heavily hunted areas of Africa (Oates, 1996), or even go extinct, as was the case for Miss Waldron’s red colobus monkey (*Procolobus badius waldroni*), which was endemic in West Africa (Oates et al., 2000). Proper game management is key to maintaining populations. For example, if hunting pressure is not excessive, adjacent undisturbed forests may provide source populations (Siren et al., 2004; Novaro et al., 2000; van Vliet et al., 2010).

Although traditional knowledge sometimes guides sustainable management and use of NTFPs (Parrotta and Trosper, 2012), NTFP harvesting can have significant adverse impacts on forest ecosystems (Belcher and Schreckenber, 2007). For example, planting and tending the saplings of benzoin trees (*Styrax* spp., tapped for resin) in the understorey of montane forests in Sumatra led, over time, to species-poor tree canopies (García-Fernández et al., 2003), and bamboo production can displace natural forests (Fu and Yang, 2004).

2.5.4 Impacts of deforestation and forest degradation on other ecosystem services

Tropical deforestation leads to complex responses of the biophysical system. Change in land cover from forest to non-forest vegetation increases albedo (i.e., the proportion of solar radiation that is reflected back to the atmosphere). However, deforestation also results in lower evapotranspiration and sensible heat fluxes, resulting in increased surface temperatures and regional reductions in precipitation (Bala et al., 2007; Werth and Avissar, 2004). Deforestation followed by conversion to grassland or cropland, and the associated changes in surface characteristics towards lighter colours, can have a strong impact on changes in albedo. Secondary forest following logging has initially higher dry-season albedo but it declines quickly and within 30 years is indistinguishable from that of the original forest (Giambelluca, 2002). Forest degradation impacts on albedo changes from selective logging may be small if a forest canopy is maintained (Miller et al., 2011). Therefore, the combined impacts of tropical deforestation on the carbon cycle and on biophysical processes (albedo and evapotranspiration) contribute to warming, locally and globally (Henderson-Sellers et al., 1993; Bala et al., 2007; Field et al., 2007). Field et al. (2007) estimated that if the average biophysical forcing from loss of tropical rainforest is 5 watts m⁻² locally, then the loss of 60 percent of global tropical forest area by 2100 would produce additional warming comparable to an extra 12 Pg C in the atmosphere.

Forests retain moisture from rainfall, allowing recharge of water tables and regulating stream flow. Deforestation and forest degradation typically result in increased soil erosion and sediment loads in streams and rivers, disrupting aquatic systems (Foley et al., 2007). Deterioration of

soil fertility, associated with deforestation and degradation pressures, increases the difficulty and cost of restoring forests or growing crops (e.g., Lal, 2005).

Furthermore, the loss of habitats and species can cause dramatic changes in trophic structure and food chains (Wright et al., 2007; Dobson et al., 2006) affecting the provision of ecosystem services mediated by species from different trophic levels; losses from higher trophic levels can trigger a cascade of unexpected effects, such as increased herbivory (Pace et al., 1999; Terborgh and Estes, 2010; Estes et al., 2011). Modifications to trophic interactions can affect key ecosystem functions and services, such as pollination and pest control (Tylianakis et al., 2007), leading to reduced production (including agricultural) and vulnerability to invasion (Laurance et al., 2006; Chapin et al., 2000). A global meta-analysis (Hooper et al., 2012) suggested that loss of plant diversity, especially of key functional species (Diaz and Cabido, 2001), can reduce plant production and decomposition rates (e.g., Cardinale et al., 2011), two key biological processes that influence carbon cycling and provisioning services.

2.5.5 Recovery of forest carbon and biodiversity following deforestation and forest degradation

Recovery of carbon and forest biodiversity following deforestation

The rates of carbon accumulation in above-ground biomass are typically fastest in the first two decades of (sub-) tropical forest succession, although it may take decades for stocks to recover to levels in primary forest (Silver et al., 2000; Feldpausch et al., 2004). Most rapid rates are in tropical rainforests and lowest in dry forests (Silver et al., 2000). Net primary productivity in secondary forests is usually three to five times greater than that of intact forests, but total carbon stocks in secondary forests are lower than in primary forests (Luyssaert et al., 2008; Lewis et al., 2009). Nevertheless, at a global scale, secondary forests are an important carbon sink that partially compensates for carbon emissions from tropical deforestation (Houghton et al., 2000; Feldpausch et al. 2005; Pan et al., 2011).

Previous land-use practices and their intensity are strong determinants of biomass recovery potential (Fearnside and Guimaraes, 1996; Steininger, 2000). Long periods of intensive use significantly impede vegetation growth (Uhl et al., 1981; Uhl, 1987; d'Oliveira et al., 2011). On a rainforest landscape in Borneo under shifting cultivation for over 200 years, biomass accumulation was significantly lower in sites cultivated six times or more because of a loss in regenerative capacity from seed-banks and of re-sprouting species (Lawrence et al., 2005). Because burning reduces stocks of available nutrients (Holscher et al., 1997; Davidson and Artaxo, 2004), repeated and shorter slash-and-burn cycles can result in progressive nutrient loss and limit capacity for biomass recovery. However, in a former rainforest area of Madagascar, five to seven cropping cycles were found to be sufficient to lead to severe degradation, because of invasion

and conversion to exotic grasslands that cannot sustain agriculture (Styger et al., 2007). These results suggest that the time period for damage and recovery varies considerably both within and among forest types.

Along with carbon recovery in tropical secondary forests, a certain proportion of primary forest species is recovered over time (Chazdon et al., 2009; Putz et al., 2012). Forest carbon and biodiversity in tropical secondary forests appear to be positively correlated over the recovery period, at least on a pan-tropical scale, but recovery of biodiversity typically lags behind that of carbon stocks (Guariguata et al., 1997; Chazdon et al., 2009; Gardner et al., 2009; Putz et al., 2012). The retention or management of structurally and floristically complex habitats, like some agroforests, can often ensure the persistence of some forest species in managed landscapes (Chazdon, 2003; Lamb et al., 2005; Scales and Marsden, 2008). Nevertheless, chronosequence studies of regenerating forests demonstrate that biotic recovery occurs over long time scales and that re-establishment of certain species and functional groups can take a century or longer (de Walt et al., 2003; Liebsch et al., 2008). Knowledge gaps remain because of limited long-term data on the recovery of secondary forests across the (sub-)tropics.

Forest structure and composition change continually as a result of disturbances, and natural successional pathways result in shifts in species and their densities over time. However, in highly disturbed forests, the species composition often differs markedly from that expected under natural processes, signifying that the ecosystem state has been altered (Aide et al., 2000; Pascarella et al., 2000). Commonly, fast growing and light-wooded pioneer species that usually only occupy small canopy gaps in primary forests dominate young secondary forests. In highly disturbed landscapes, natural successional processes may be arrested and invasive species, if present in the landscape, may become dominant in the forest canopy (Grau et al., 2003; Lugo, 2002; Chazdon, 2008; Letcher and Chazdon, 2009) resulting in changes in ecological processes, and often the loss of some ecosystem services (e.g., van Wilgen et al., 2001).

Many studies report substantial recovery of biodiversity in secondary forests following slash-and-burn cultivation (e.g., Raman, 2001; Dunn, 2004; Chazdon et al., 2009). The capacity of forests to recover biodiversity during the fallow phase depends, in part, on the duration and intensity of agricultural management. Lawrence (2004) showed a long term erosion of tree diversity resulting from shifting cultivation in Borneo, where the evenness of the tree community declined with each cultivation cycle. A study in rainforests and dry forests in Tanzania comparing the recovery from slash-and-burn agriculture across areas with different fallow periods showed that forest recovery was higher for both biomass and tree diversity after long fallow periods, and recovery occurred only where the cultivation period was less than 16 years (Mwampamba and Schwartz, 2011). A review of 65 studies by Dent and Wright (2009) found that secondary forests resulting from low intensity management systems, such as shifting cultivation, appeared to be more similar

to primary forests than those regenerating from pastures or intensive agriculture. Nevertheless, caution is needed in interpreting these latter results because few taxa were studied, and they often lacked proper spatial coverage, sufficient replication or appropriate controls (Gardner et al., 2007; Lewis, 2009)

Recovery of carbon and forest biodiversity after forest degradation

The consequences and recovery times for forest carbon depend on the level, scale and forms of degradation. In some cases, recovery from degraded alternative stable states is not possible without substantial management (e.g., van Wilgen et al., 2001). Repeated burning has a strong detrimental effect on carbon accumulation in secondary forests, with five or more burnings reducing total carbon accumulation by over 50 percent (Zarin et al., 2005). In areas subject to excessive burning, secondary forests are unable to recover their original biomass within the average fire-return interval for several reasons: continuing post-fire mortality (Baker et al., 2008; Barlow et al., 2003; 2010), reduced carbon accumulation rates with repeated fires (Zarin et al., 2005) and changes in species composition in burned forests towards short-lived, fast-growing species with low wood density (Barlow and Peres, 2008; Cochrane and Schulze, 1999; Slik et al., 2010; Slik et al., 2002).

Some forest species recover slowly over time after fire. For example, the avian species composition of burned forests can become less similar to that found in unburned forests over time due to lag-effects in biodiversity responses (Adeney et al., 2006; Barlow and Peres, 2004). Shifts in species composition following timber extraction can either be temporary (e.g. bats in Trinidad; Clarke et al., 2005) or persist for decades or more (e.g., ants in Sri Lanka; Gunawardene et al., 2010 or plants in India; Devi and Behera, 2003). Ecosystem recovery following logging also depends on the methods used (Asner et al., 2004), and differences in the condition of the forest prior to logging can have a dramatic impact on trajectories of ecological recovery. Bischoff et al. (2005), working in Borneo, found that logging late successional forest characterised by dense understorey pioneer vegetation, after disturbance over 100 years ago, resulted in an increase in shade-tolerant and small stature tree species at the expense of canopy hardwood species. There is some indication of greater levels of biotic resilience to logging in forests that are regularly exposed to natural disturbances (e.g. fire and hurricanes in Belize; Whitman et al., 1998) or regions that underwent a relatively rapid expansion and contraction of forest areas during the Pleistocene (e.g. West Africa; Ernst et al., 2006). A lack of long-term data on the recovery of secondary forests across the (sub-)tropics means that the time required to recover the forest biota to a desirable state remains highly uncertain.

The emergence of novel (sub-)tropical forests

Accumulating human impacts with consequent cascading effects on biological processes and unpredictable stochastic effects combine to generate ecological conditions and

species interactions that have no evolutionary precedents (e.g., Tylianakis, 2009). These degraded forests and recovering deforested areas have resulted in novel ecosystems globally (e.g., van Wilgen et al., 2001; Hobbs et al., 2006; Lindenmayer et al., 2008). Such new species assemblages may or may not provide all of the goods and services that humans need because often original dominant functional species have been lost or reduced in numbers, with the commensurate alteration of processes (e.g., Olden et al., 2004; Lewis, 2009).

There is growing evidence to suggest that the rate of many ecological processes may be both magnified and accelerated in modified tropical forest landscapes, with unpredictable implications for the maintenance of biodiversity (Lewis, 2009; Laurance et al., 2002). Novel systems may foster new patterns of species loss as extinction is most likely to occur when new threats or combinations of threats emerge that are outside the evolutionary experience of species, or threats occur at a rate that outpaces adaptation (Brook et al., 2008). However, novel systems can also provide important refuges for recovering forest biodiversity in areas that have been reforested or highly degraded. For example, in Puerto Rico, the naturalisation of introduced tree species on abandoned agricultural lands is thought to have played an important role in the recovery of many native species (Lugo and Helmer, 2004).

Although the definition of what constitutes a 'novel ecosystem' remains somewhat arbitrary, their emergence follows the selective loss and gain of key taxa, the creation of dispersal and regeneration barriers, or changes in system productivity that fundamentally alters the relative abundance and structure of resident biota (Hobbs et al., 2006). Two compelling examples are the creation of 'new forests' in Puerto Rico that are comprised of species assemblages that have not previously been recorded (Lugo and Helmer, 2004), and the major alteration of the structure of native Hawaiian rainforests following the naturalisation of numerous alien invasive plants (Asner et al., 2008).

Understanding the structure and function of novel ecosystems is of fundamental importance in evaluating patterns of biodiversity change, and the potential for biodiversity recovery in degraded areas (Chazdon, 2008).

2.6 Conclusions

1. Biodiversity maintains critical ecosystem processes (e.g., seed dispersal, photosynthesis) and underpins the provision of many forest ecosystem services. For some ecosystem services, such as pollination, there is a direct link between species richness and the provision of the service. Other ecosystem services, such as erosion control, are largely independent of species composition and richness. Larger areas of forest are essential for the provision of many ecosystem services (e.g. carbon storage, climate regulation), are more productive and deliver more services than small forest patches because of higher biodiversity, reduced edge effects and less human access. Small forest patches do maintain some services in highly modified landscapes.

2. Forests contain most of Earth's terrestrial biodiversity, especially in (sub-)tropical rainforest, moist forest and montane systems. Deforestation and forest degradation within these ecological zones are the largest drivers of terrestrial biodiversity decline.
 3. Very extensive areas of degraded forests and of deforested areas now exist in tropical and sub-tropical regions, with significant adverse effects on conservation of carbon stocks and biodiversity, and thus on the provision of many ecosystem services. The combined impacts of past and ongoing degradation on forest carbon and biodiversity may approach those of deforestation.
 4. Ecosystems can exist in various states, but not all states provide the same level of ecosystem services. Human-induced losses of biological diversity can adversely affect the resilience of forest ecosystems, and hence the long-term provision of services. To avoid catastrophic change, managers need to ensure that ecosystems remain within a 'safe operating space'.
 5. In forests with few tree species such as most planted forests, increases in tree species richness may lead to increased biomass carbon stocks and some other services under appropriate conditions. At high levels of diversity, the relationships between changes in species richness and production remain poorly understood.
 6. Different forest types and ages can vary markedly in levels of biodiversity and the amount of carbon stored in different pools; however primary forests store high levels, while young forests sequester carbon rapidly. Accordingly land use planning processes need to take these differences into account when addressing both biodiversity and carbon objectives.
 7. In (sub-)tropical forests recovering from major disturbance, carbon and biodiversity both increase over time, but recovery rates for both diminish over time, and recovery of biodiversity is typically much slower than that of carbon.
 8. Due to the large number of endemic species, endangered species, and unique species assemblages in (sub-)tropical forests, spatial planning for biodiversity conservation objectives needs to be more area specific than is necessary for carbon management.
 9. There is uncertainty with respect to the capacity of 'novel' forest ecosystems, which differ in composition and/or function from past systems as a consequence of changing species distributions and environmental alteration, to provide expected goods and services in future.
- The majority of scientific studies have not distinguished the effects of species composition on forest productivity and other ecosystem functions, from the effects of either species richness or of individual species. Further work is needed to better understand: (i) the relationships between plant species richness and functional diversity, and biomass accumulation in diverse forest systems, especially for novel systems; (ii) the relationships between species richness and ecosystem resistance (to chronic disturbances); (iii) the cascading effects of the loss on faunal diversity on forest ecosystem processes; (iv) the long-term effects of repeated degradation events on rates of recovery of forest ecosystems; (v) the existence of degradation/disturbance thresholds or tipping points beyond which recovery of expected biodiversity, ecosystem functions and provision of services is severely constrained.
 - Further work is needed to improve our knowledge of the magnitude and dynamics of below-ground carbon stocks and fluxes in different forest types. The time scales and conditions required to recover to pre-disturbance levels of biodiversity and carbon in secondary forests (which are of significant value to conservation of both carbon and biodiversity), are poorly understood. There is also considerable uncertainty regarding the levels of ecosystem service provision from increasingly widespread novel ecosystems that result from prolonged anthropogenic impacts.

Significant gaps exist in our understanding of the relationships between biodiversity and ecosystem functioning and provision of forest ecosystem services, including carbon sequestration, and how these relationships are affected by forest condition.

Selected studies in all types of forests on the relationship between species richness and carbon fluxes and storage

Ecological Zone	Location	Forest Stand Type	No. Tree Species	Age (y)	Carbon pools	Carbon Fluxes	Type of relationship Biodiversity vs Carbon	Positive (+) Negative (-) Neutral (0)	Reference
Sub-tropical moist	Puerto Rico	Planted (experimental)	1 to 4	8.5	Soil, above-ground biomass	Net Primary Production (NPP), decomposition	Carbon increase in mixed species plantations	+	Parrotta (1999)
Sub-tropical rainforest	Costa Rica	Planted (experimental)	1, 8	5	Above-ground biomass	NPP	Biomass greater in mixed plantations than in monocultures	+	Montagnini (2000)
Sub-tropical	Costa Rica	Planted (experimental)	1 to 3	13	Above-ground biomass	NPP	Higher growth rates in mixed plantations	+	Petit and Montagnini (2004)
Sub-tropical rainforest	Puerto Rico	Natural	50+	multi	Above-ground biomass	NPP	Positive relationship between biodiversity and AGB but varied in strength with the indicator and age of stand.	+	Vance-Chalcraft et al. (2010)
Tropical humid	Australia	Planted (commercial)	1 to 8		Above-ground biomass	NPP	Higher tree growth rates with increase sp. richness	+	Ersikine et al. (2006)
Tropical rainforest	Australia	Planted (commercial)	1 vs. 2	10	Above-ground biomass	NPP	Two species resulted in superior growth rates than either species alone.	+	Bristow et al. (2006)
Tropical rainforest	Panama	Planted (experimental)	6	6	Litter	Litter decomposition	Significant effects of tree species richness on total litter production; litter decomposition not affected by tree species richness	+	Scherer-Lorenzen et al. (2007)
Tropical rainforest	Australia	Planted (experimental) & Natural	2 to multi	23-72	Above-ground biomass	NPP	No relationship between diversity and production in old forests (ie, possible age effect)	0	Firn et al. (2007)
Tropical rainforest	Panama	Natural	multi		Above-ground biomass	NPP	No relationship between tree-species diversity and above-ground carbon stocks	0	Kirby and Potvin (2007)
Tropical rainforest	Panama	Planted (experimental)	1 vs 2 (total 6)	3	Above-ground biomass	NPP	Tree biomass higher in pairs and plots with higher species richness	+	Murphy et al. (2008)
Tropical rainforest	Panama	Planted (experimental)	1 vs 2 (total 6)	3	Belowground biomass		No significant differences in root or microbial biomass	0	Murphy et al. (2008)
Tropical rainforest	Panama	Planted (experimental)	1 vs 2 (total 6)	3	Soil	Soil respiration	Soil respiration 19-31% higher in monoculture than in pairs and plots with higher species richness	-	Murphy et al. (2008)
Various (Review)	Tropics	Planted (experimental)	2 -4	1.5-14	Above-ground biomass	NPP	Mixed plantations had higher diameter growth rate	+	Piotta (2008)
Tropical rainforest	Panama	Planted (experimental)	1,3,6	4,5	Above-ground biomass	NPP	Species diversity explained 23% of productivity and 30% of mortality	+	Healy et al. (2008)
Tropical rainforest	Borneo	Natural			Soil		Tree diversity negatively correlated with organic carbon content	-	Slik et al. (2009)

Tropical rainforest	Brazil	Planted (experimental)	2, 4 or 7		Litter	Litter decomposition	Species composition, but not species richness, significantly influenced litter decomposition rates	0	Gieseimann et al. (2010)
Tropical rainforest	Panama	Natural	30-61	~200	Above-ground biomass		Species richness increases tree carbon storage	+	Ruiz-Jaen and Potvin (2010)
Tropical rainforest	French Guyana	Planted (experimental)	16		Litter	Litter decomposition	Effects on decomposition mostly driven by species composition; functional litter diversity in chemical traits did not explain decomposition	0	Barantal et al. (2011)
Tropical rainforest	Panama	Planted (experimental)	6	8	Litter; Soil	CWD decomposition Litter decomposition; Soil respiration	Significant effects of diversity on CWD decomposition and soil respiration	+	Potvin. et al. (2011)
Tropical rainforest	Panama	Planted (experimental)	6	8	Soil Below-ground biomass		No diversity effects on the carbon pools No significant differences in root or microbial biomass	0	Potvin. et al. (2011)
Tropical rainforest	Panama	Planted (experimental) & Natural	6		Above-ground biomass		Tree carbon storage in the mixed-species plantation was mainly explained by species richness and in the natural forests by functional trait diversity	+	Ruiz-Jaen and Potvin (2011)
Temperate	Southwest China	Natural	48, 80, 143 & 310		Above-ground biomass		No relationship species richness and above-ground carbon; significant negative relationship between species diversity and above-ground carbon storage	0 -	Zhang et al. (2011)
Boreal	Sweden	Natural			Above- and below-ground biomass		Vascular plant richness and diversity was positively correlated to total carbon storage; negatively correlated with above-ground C storage; positively correlated to below-ground carbon storage	+	Wardle et al. (2012)
Temperate and tropical	Global	Model, Natural			Above-ground biomass	NPP	No relationship at very large scales between tree species richness and biomass/ha	-	Enquist and Niklas (2001)
Temperate	USA	Survey data			Above-ground biomass	NPP	Positive relationship between tree biomass and species richness across USA	+	Caspersen and Pacala (2001)
Temperate	Germany	Planted (experimental)			Above-ground biomass	NPP	Positive relationship between species richness and tree biomass	+	Pretzsch (2005)
Temperate	UK	Planted (experimental)			Above-ground biomass	NPP	Positive relationship between species richness and tree biomass	+	Jones et al. (2005)
Temperate & Boreal	Canada	Planted (experimental) & Natural	1 to 13		Above-ground biomass	NPP	Positive relationship in temperate forests and in boreal forests	+	Paquette and Messier (2011)

Forest area and above-ground biomass (AGB) carbon and total biomass carbon by region and ecological zone in the tropical and sub-tropical domain. Based on the Saatchi et al. (2011) tropical biomass maps*, the FAO ecological zones (Figure 1.1; FAO, 2001) and the MODIS forest cover map (25 percent forest cover threshold) (NASA, 2010). Carbon is defined as 50 percent biomass. SD is standard deviation of spatial variations of the estimates across the regions; SE represents standard errors of the estimates, i.e. uncertainties associated with all errors from data sources and estimation methods

Region	Forest Area (Mha)	AGB carbon density (Mg C ha ⁻¹)		Total biomass carbon density (Mg C ha ⁻¹)	
		Mean ± SD	SE (±)	Mean ± SD	SE (±)
Africa					
Tropical rainforest	252.9	107 ±51	37	135 ±64	47
Tropical moist deciduous forest	110.6	38 ±18	13	50 ±23	17
Tropical shrubland	1.6	41 ±25	14	53 ±32	17
Tropical dry forest	36.1	38 ±18	13	49 ±23	16
Tropical mountain system	22.7	64 ±39	22	82 ±49	28
Sub-tropical humid forest	1.5	38 ±15	13	49 ±19	17
Sub-tropical dry forest	0.7	31 ±16	11	41 ±21	14
Sub-tropical mountain system	1.1	34 ±11	11	45 ±14	15
Africa Total	427.2	80 ±78	27	102 ±98	35
Latin America					
Tropical rainforest	587.1	115 ±34	37	146 ±42	47
Tropical moist deciduous forest	179.3	54 ±42	18	69 ±53	24
Tropical shrubland	0.9	55 ±41	19	71 ±51	24
Tropical dry forest	47.6	27 ±23	10	36 ±29	13
Tropical mountain system	71.8	86 ±50	29	110 ±62	37
Sub-tropical humid forest	20.4	51 ±38	18	66 ±48	23
Sub-tropical dry forest	5.3	55 ±51	20	71 ±64	25
Sub-tropical mountain system	7.2	21 ±23	8	27 ±29	10
Latin American Total	919.8	94 ±110	31	119 ±138	39
Southeast Asia					
Tropical rainforest	261.6	121 ±50	44	153 ±62	56
Tropical moist deciduous forest	55.6	105 ±49	37	133 ±61	47
Tropical shrubland	2.5	64 ±39	23	82 ±49	29
Tropical dry forest	17.6	83 ±50	30	106 ±63	38
Tropical mountain system	53.6	128 ±34	43	162 ±42	55
Sub-tropical humid forest	0.8	88 ±34	32	112 ±42	41
Sub-tropical mountain system	7.7	101 ±41	37	128 ±52	47
Southeast Asia Total	399.5	118 ±114	42	149 ±142	54
All Total	1,746.5	96 ±177	32	122 ±221	41

* AGB was mapped using a combination of data from in-situ inventory plots and satellite light detection and ranging (LIDAR) samples of forest structure, plus optical and microwave imagery. Below-ground biomass (BGB) was calculated as a function of AGB (BGB = 0.489 AGB^{0.89}) with total biomass = AGB+BGB (for more information see Saatchi et al., 2011)

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Chapter 3

Impacts of forest and land management on biodiversity and carbon

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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; Svancara 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

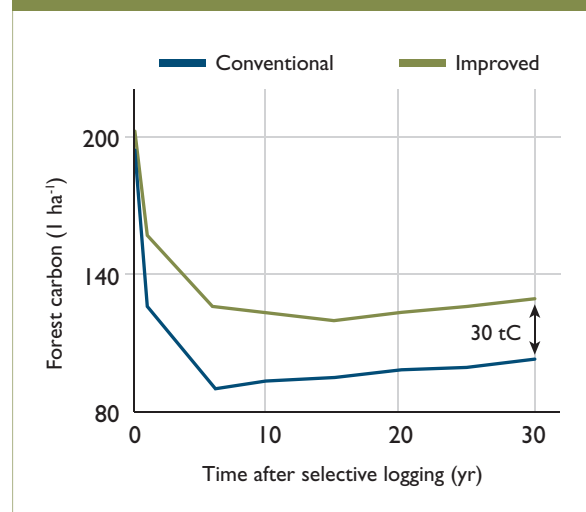
The International Tropical Timber Organization (ITTO; Blaser et al., 2011) estimates the size of the natural tropical permanent forest estate (PFE) to be about 760 M ha, comprising 400 M ha of production forest and 360 M ha of protection forest (designated to remain as forest for purposes other than 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.



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

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

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; Kelty, 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 (Brockerhoff 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 (Brockerhoff 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; Brockerhoff 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; Brockerhoff 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* (Brockerhoff 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 (Brockerhoff 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.

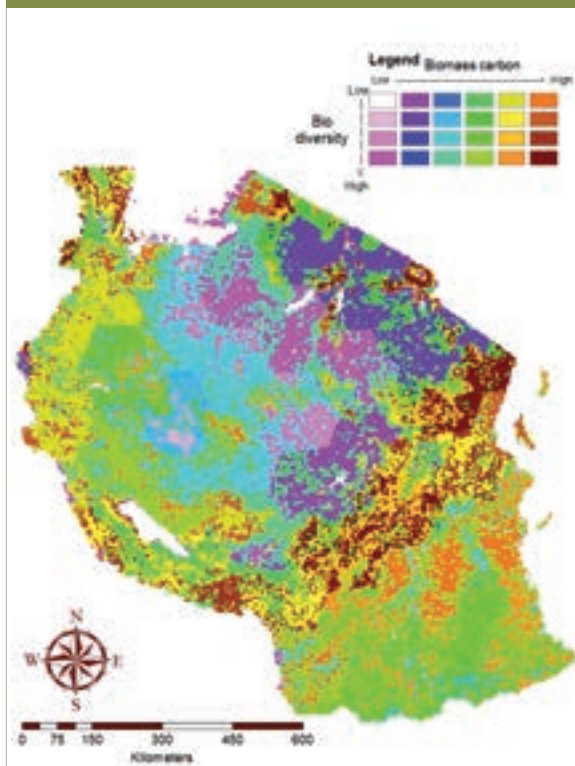
Biodiversity poses a particular challenge for land-use planning and management because the composition of species and habitat types can vary greatly from place to place. Unlike carbon, for which stocks and fluxes can

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).

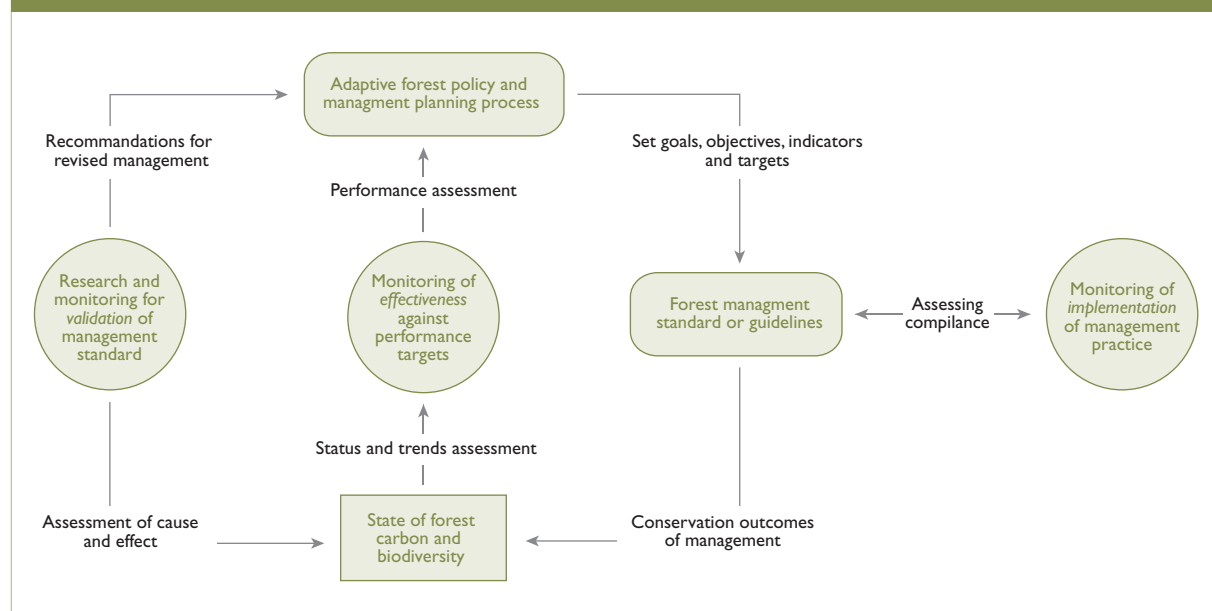
3.4 Monitoring to assess changes in forest carbon and biodiversity

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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Chapter 4

Social and economic considerations relevant to REDD+

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Abstract: While REDD+ has the potential to generate substantial positive impacts for climate mitigation and biodiversity, the way in which it is implemented will determine the extent to which it is associated with positive social and economic outcomes. The primary objectives of REDD+, avoiding deforestation and forest degradation, can greatly benefit poor people, who are often disproportionately impacted by the loss of forests and the services they provide. Further, if a significant fraction of likely financial flows associated with REDD+ directly or indirectly reaches the rural poor, it might generate considerable benefits. On the other hand, the poor are also most vulnerable to changes in regimes for resource management and access that may be associated with REDD+, with severe negative consequences for their already marginal livelihoods. If REDD+ is to address the social, political and economic factors that produce inequitable outcomes, it must give parity to socio-economic objectives alongside carbon and biodiversity goals. Equally, the most adverse social and economic consequences need to be avoided through the adoption of strong safeguards, which are sensitive to, and include monitoring systems for, tracking social impacts. Pursuing these social objectives alongside REDD+ is likely to not only make the process more equitable but also increase the likelihood of achieving carbon and biodiversity goals. However, it is important to recognise that 'win-win' outcomes are not always possible and there are sometimes difficult trade-offs to be negotiated between carbon, biodiversity and social objectives. Integrated landscape management provides a powerful tool to address and reconcile the many environmental, social and economic aspects relevant to REDD+ inside and outside forests. Careful and inclusive (participatory) spatial planning can positively influence the distribution of winners and losers across the landscape so that REDD+ acts in the interests of the most vulnerable groups, thereby resulting in positive impacts on both equity and environmental effectiveness.

4.1 Introduction

All REDD+ interventions - which typically attempt to alter incentives for the management of *forests*¹, the structures under which they are governed, or the drivers of forest loss - are inevitably associated with differentiated impacts on individuals, households and communities as well as the wider economic relations within which these groups are embedded. This chapter discusses the implications of these socio-economic trade-offs (and synergies) for the implementation, effectiveness and sustainability of REDD+.

Whilst it is still premature to evaluate the impacts of REDD+, lessons can be learned from other related forest interventions which may inform REDD+ implementation. One certainty is that any forest conservation or management intervention will generate social and economic impacts, and the poorest are often the most vulnerable to negative impacts. Research in this context has highlighted the diverse and complex ways in which people benefit from forests, and the entrenched political and economic asymmetries that pervade forest conservation interventions. Given that REDD+ is likely to be implemented in this socio-economic and political landscape, it is important to understand the social and economic consequences of earlier strategies, and to learn lessons from these strategies for REDD+.

There are a number of reasons for concern about the potential social and economic impacts of REDD+. Perhaps most importantly, there is a strong moral and ethical case to care about the well-being of those who are potentially negatively affected by any proposed change in *forest management* systems. Impacts are felt not just at the

individual level, but also by groups and communities, and it is necessary to consider who gains and who loses when proposed interventions are implemented. As far as possible, direct negative impacts felt by stakeholders from local to global scales need to be avoided, or mitigated. Moreover, negatively affected individuals and groups might threaten the viability and likely effectiveness of REDD+ interventions, providing a further, more instrumental justification for a focus on economic and social impacts.

The services that natural *ecosystems* provide have been estimated to have global aggregate values in the trillions of dollars per year (Costanza et al., 1997; Balmford et al., 2002; TEEB, 2008). Estimates suggest that the economic value of the climate change *mitigation* potential from REDD+ alone is likely to exceed the aggregate economic losses that would stem from foregone agricultural production (Stern, 2007; Eliasch, 2008; Strassburg et al., 2009; TEEB, 2009). The potential for REDD+ arises from comparing these high aggregate global values with the estimated opportunity costs of the foregone agricultural production or, perhaps more appropriately, with the costs of changing local land and resource use practices in target areas.

On an aggregate level, although REDD+ interventions have considerable potential for positive socio-economic outcomes, these interventions need to be managed under very specific conditions to ensure that they do not adversely affect particular groups. As will be seen in detail in the remainder of this chapter, the challenges of generating positive outcomes where losers are properly compensated are particularly complex in the case of forest-related interventions. Any intervention carries the potential

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

for unintended consequences, but these can be particularly severe in the context of rural areas in developing countries where weak institutions, poor *governance* and under-developed markets are often the rule, and existing structures for forest governance and resource management tend to be biased against the interests of poor and marginal forest-dwelling and indigenous communities. In light of the complex, dynamic relationship between forest conservation and human well-being, it is very difficult to consistently align the needs of local people with those of conservation. This experience challenges the idea that optimal or 'win-win' interventions will soon be the norm, for example under REDD+, as this idea is simply not borne out by any rigorous evaluation of long-term *forest management* histories in the tropics.

The chapter starts by highlighting some over-arching considerations and mediating conditions that are important for an understanding of the social and economic impacts of forest management decisions. This is followed in Section 4.3 by a brief review of the social and economic impacts of current patterns of *deforestation* and *forest degradation*, paying specific attention to the evidence of impacts on the most vulnerable groups. Section 4.4 adopts a landscape approach to examine lessons from interventions which influence agricultural drivers, review the experience of protected area management as well as strategies to reduce the impacts of extractive use from forests, and consider the social and economic impacts of previous *forest restoration*, *reforestation* and *afforestation* activities. These interventions all provide valuable lessons for REDD+, highlighting the types of behavioural, institutional, governance and socio-economic changes that these have involved, and the outcomes that have been observed, in terms of equity, efficiency and effectiveness. This is followed by a section which focuses on major approaches that have been used to implement these four management actions, focusing in particular on the role of decentralisation and participatory forms of forest governance and management, and also considering ways in which interventions attempt to incentivise behavioural change for stakeholders, drawing particularly on experience with *payments for ecosystem* (or *environmental*) *services* (PES) schemes, and forest certification. Section 4.6 considers the implications of these previous forest-related interventions for strategies that seek to find synergies between reductions in *greenhouse gas* emissions, improvements in *biodiversity*, and positive social and economic outcomes. This final section also analyses lessons learnt and raises some key issues for REDD+ strategies, which are discussed in greater detail in Chapter 5.

4.2 The social and economic context for forest management

Forests provide inputs to the well-being of people and societies, especially for forest-dwelling peoples, both in terms of the material ('economic') dimensions of everyday livelihood activities (which include subsistence, income generation, shelter, employment and trade), as well as non-material ('social') aspects, which include cultural

and spiritual values, quality of life, health and well-being and more fundamental issues related to identity, aspirations, political systems and human rights (MA, 2005).

Forest management interventions, and restrictions on the use of forest resources, typically have differentiated impacts on stakeholders, with identifiable patterns of winners and losers. Changes which might result in aggregate improvements in the quality and quantity of forest resources, as well as their long term sustainability, may be associated with significant negative impacts on the social and economic well-being of particular individuals and groups (Chan et al., 2007). People relate to, and benefit from, forests in diverse and complex ways, which make 'win-win' solutions difficult to identify or to sustain. Impacts might manifest themselves at different scales, for instance, the beneficiaries from certain forms of conservation and sustainable use activity may be at national or global scales, while the losers are likely to be those who live more locally (Fearnside, 2003). Social impacts include changes to people's ways of life, their culture, communities and political systems, as well as their surrounding environment, health and well-being, rights, and fears and aspirations (Vanclay, 2003). Individuals and groups who suffer these adverse consequences are often politically weak and therefore either neglected from consideration, or inadequately compensated for their losses. Of particular concern in this context, are unequal gendered experiences of forest access, use and management (Meinzen-Dick et al., 1997; Agarwal, 2010). Concerns over the lack of justice in forest interventions frequently arise from observing these inequitable outcomes, and their socially differentiated consequences (Corbera et al., 2007; McDermott and Schreckenberg, 2009).

Where forest interventions create benefits for local populations, these are often inequitably distributed across existing social fault-lines of wealth, ethnicity and gender. Furthermore, literature shows that such inequitable distribution is highly resilient to change, with attempts to improve well-being for the poorest frequently resulting in elite capture of most of the benefits (Jumbe and Angelsen, 2006; Blom et al., 2010; Iversen et al., 2006; Naughton-Treves et al., 2011). This is true of strictly protected areas (Ferraro, 2002; Shyamsundar, 1996; Nautiyal and Nidamanuri, 2012), forms of devolved and collective forest management (Naidu, 2011; Kusters et al., 2008; Jagger, 2008) and is emerging as an issue in market-based schemes (McAfee and Shapiro, 2010; To et al., 2012; Corbera and Brown, 2010; Lansing, 2011). Such inequity is undesirable in itself (the moral case), and is also thought to undermine long-term effectiveness of biodiversity conservation or *carbon storage* (the instrumental case) because it fails to incentivise conservation-oriented behaviours.

The relationships between people and forests are typically mediated through specific structures which control, manage, exclude and privilege certain types of forest use, with unequal consequences for differentiated groups of social actors. Important mediating factors include: structures of governance and the exercise of authority; the nature of rules and institutions for resource management;

as well as types of tenure and property rights regimes which prevail in any particular context. These themes will recur throughout this chapter, and are introduced briefly in this section.

4.2.1 Governance and the exercise of authority

Governance can be understood as any attempt to coordinate human actions, usually directed towards particular goals. The concept is relevant at multiple scales, from the coordination of activities within households, which govern, for instance, the allocation of labour to forest-related tasks, to the international context, where forest governance typically involves the ways in which state and non-state actors negotiate and collaborate over forest management principles and norms, as well as rules and decision-making procedures (Brown, 2001; Agrawal et al., 2008; see Chapter 5). In the context of forest governance, coordination of activities often requires the exercise of authority, usually the power to enforce rules of access and exclusion, and to punish rule violations. State actors often take on these roles, but forest-based peoples and movements frequently contest the state's monopoly on coercion, especially if the use of state authority is perceived to be biased or unfair (Ribot et al., 2006). Non-state governance regimes can often be very effective in coordinating the behaviour of stakeholders, especially at local levels, as they are seen to be more accountable, legitimate and accessible than more distant state authorities (Colfer and Capistrano, 2005; Cashore et al., 2007). Transparency and accountability are critical factors for forest governance, and for the legitimate exercise of authority (Colfer et al., 2008). A related issue is that of corruption, which potentially prevails at different levels within the forest sector, from national level clientelist networks which facilitate illegal logging and trade in illegally harvested timber, to more 'petty' corruption where lower level forest officials use their authority to extract bribes from forest users. Social structures are often very relevant to understanding how power is exercised, and it is important to understand these wider social relations to address issues such as equity, justice and fairness when considering forest governance (Sikor et al., 2010). The gendered dimensions of this are particularly important, as men and women's knowledge of, and management strategies for, forests may be distinct, and are directly related to their use and dependence on forest resources; while women's role in forest management is critical, they have often been politically and culturally marginalised from participation in decision making (Agarwal, 2010).

4.2.2 The rules and institutions which control stakeholder behaviour

The specific rules that control what forest users can and cannot do, and how these impact on their incentive structures, are a key feature of different forest management systems. Institutions provide access to information and resources, shape incentives and structure the context within

which social interactions take place (Ostrom, 1990). Definitions often distinguish between formal institutions, especially the legal framework that governs rights, and informal structures such as rules, norms and conventions (North, 1990). Rules and institutions ultimately influence the detailed ways in which stakeholders interact with each other, and with forest resources, as they determine issues such as who is allowed access and for what reasons, the extent of harvesting that might be permitted, the seasonality of use, and the restrictions on trade, transport and sale of forest products. Of particular interest, in the current context, is the role of commercial and market-like transactions for forest-based goods and services, which might be socially or legally restricted in some jurisdictions (especially where forests provide public good benefits), while increasingly prevalent in others (Pagiola et al., 2002). These detailed rules are typically enforced through a variety of mechanisms, which emerge from the wider social context within which forest use is embedded, and which provide the authority which secures the legitimacy of such rules. Rule enforcement often involves monitoring, and is closely related to the governance regime and types of property rights arrangements that prevail in any particular context (Sikor and Lund, 2009).



Women are very active forest managers in many parts of Amazonia. Photo © Miguel Pinedo-Vasquez

4.2.3 The nature of tenure and property rights

Two issues are important when considering the role of tenure and property rights in the context of forest management. The first is tenure security, which indicates the extent to which the rights of land or forest owners are recognised and protected, thereby providing them with

incentives for investment and sustainable management. In the absence of secure tenure, forest owners and managers may be forced to adopt short term strategies that do not provide effective stewardship of the resource, and are potentially detrimental to their own long term interests (Godoy et al., 1998). The second issue has to do with the type of property regime which prevails in the context of forest management (Feeny et al., 1990; Ostrom, 1990; Bromley, 1991). Ownership and control are important to secure the benefits that flow from forest resources, but there are many different property regimes which determine the ways in which these flows are appropriated and shared. Communal property systems typically recognise multiple interests, and usually require some form of collective action in order to be effective. On the other hand, private property provides secure ownership, but can often be highly inequitable because it is exclusive (Feeny et al., 1990; Bromley, 1991). Evidence also shows that individualisation of land and related resources harms women's rights (Mwangi and Mai, 2011). State-ownership is associated with varying levels of restrictions on access and use, which differently incentivise stakeholders. Participatory regimes are potentially a mechanism for promoting mutually-beneficial outcomes, frequently involving the collaboration of multiple stakeholders, often from different sectors (e.g. states and communities working together in systems of joint forest management, company-farmer partnerships as part of outgrower arrangements); however, maintaining collaboration is not always easy, especially as social and economic circumstances change (Borrini-Feyerabend, 1996; Grimble and Wellard, 1997; Kellert et al., 2000; Vira and Jeffrey, 2001).

Forest management decisions have historically been shaped by the ways in which forest use and access are controlled and governed, by the structure of rules and institutions which shape human interaction, and by the detailed tenure and property rights arrangements that prevail in specific contexts. These three factors are critical to any understanding of forest management practices, and will typically provide the over-arching context within which particular interventions might impact the social and economic well-being of individuals and communities. The discussion in this chapter analyses how these three key mediating factors influence the social and economic consequences of forest-related interventions.

4.3 The social and economic impacts of deforestation and forest degradation

Processes of deforestation and forest degradation have economic and social consequences at multiple scales, from the loss of forest-based *ecosystem services* at local, national, regional and global levels, to the contributions that forests make to national and household level economies, to the social dislocation that occurs from the loss of access to locally-valued forest resources. These impacts tend to be unevenly distributed over both space and time, and often disproportionately affect those who are most

vulnerable (Chomitz, 2007). Understanding these consequences provides additional justification for addressing the drivers of deforestation and forest degradation, as doing so potentially reverses these negative consequences for those groups that are currently most dependent on forests for their well-being.

Estimates of *forest-dependent* populations across the world vary, with the Millennium Ecosystem Assessment (MEA) suggesting that over 1.6 billion people world-wide depended to varying degrees on forests for their livelihoods at the turn of the century. The MEA also estimated that forests provide a home to almost 350 million, and about 60 million indigenous people almost wholly depend on forests (MA, 2005). In a recent attempt to compile figures from a variety of research, official and NGO sources, the Forest Peoples Programme estimated that forest-dependent population numbers lie between 1.095 billion and 1.745 billion, or between 14 and 25 percent of all humanity (FPP, 2012). Whatever the precise number, these estimates indicate that the impacts of deforestation and forest degradation could be devastating to the lives of millions, especially those who do not have the means to find alternatives to forest-based good and services.

Forest goods and services contribute directly to local livelihoods through inputs to agriculture, as products to consume and sell locally (Cavendish, 2000; Eliasch, 2008), and as inputs to wider production value chains (Stoian, 2005). In addition to their provisioning services, local people value forests for their regulation of water supplies to agriculture and to fishing (Dennis and Masozera, 2009), prevention of soil erosion (Yaron and Moyini, 2003) and moderation of local climatic conditions perceived as beneficial to agriculture and health (Hartter, 2010; Berbés-Blázquez, 2012). Forests are also important for their recreational, cultural and spiritual values; and they also play a role in insurance and risk spreading for forest-dependent communities.

Households that live in and around forests are estimated to derive significant proportions of their annual income from forest resources (see for instance, CIFOR's Poverty and Environment Network, and studies reviewed in Vira and Kontoleon, 2010). The exact figures vary, but some recent work on the extent of household income derived from forests or *non-timber forest products* (NTFPs) is summarised in Table 4.1, below.

The evidence also suggests that forest-dependence is widespread globally, with many studies reporting that a majority of rural households are engaged in some form of forest-based livelihood activity (Levang and Dounias, 2005; Shackleton and Shackleton, 2006; Dovie and Shackleton, 2007; Jha, 2009; Sharma and Gairola, 2009). Some empirical work demonstrates the unequal nature of this dependence, with a number of studies suggesting that the poor are disproportionately dependent on forest resources (for instance, Reddy and Chakravarty 1999; Levang and Dounias, 2005; Babulo and Muys, 2008; Sapkota and Oden, 2008; Kamanga and Vedeld, 2009). Poorer households tend to rely more on forests to meet their basic subsistence needs (Ravi and Bull, 2011; Coomes et al., 2011) or to generate income (Bush, 2009;

Evidence of forest-derived income

Table
4.1

Source	Region	Percent of household income derived from forests
Bahuguna, 2000	South Asia	48.7% of household income
Fisher, 2004	Southern Africa	30% of household income
Fu and Chen, 2009	China	1.7% of household income in commercial plantation system, 12.2% in subsistence system
Kamanga and Vedeld, 2009	Southern Africa	15% of total household income
Levang and Dounias, 2005	South-east Asia	30% of total household income
Mamo and Sjaastad, 2007	East Africa	39% of total household income
Shaanker and Ganeshiah, 2004	South Asia	Between 16%-59% of household income in three different sites
Viet Quang and Anh, 2006	South-east Asia	For 30% of households, over 50% of total income; further 15%, 25-50% of total income

Angelsen and Kaimowitz, 1999; Rugendyke and Son, 2005), whereas relatively wealthy households use forests more to supplement their incomes (Bush, 2009; Ravi and Bull, 2011; Fisher and Shively, 2005; Adhikari and Di Falco, 2004; Narain and Gupta, 2008; Coulibaly-Lingani and Tigabu, 2009). For example, Fisher (2004), drawing on research in Southern Africa, found that while the poor are disproportionately dependent on low-return forest activities which feed into subsistence strategies, wealthier households depend more on high-return forest activities, typically associated with income generation and the sale and trade of forest products and services. However, it is important to be cautious about how evidence on the disproportionate forest dependence of the poor is interpreted, as this may potentially reflect a 'poverty trap' (Angelsen and Wunder, 2003). While these groups may appear more linked to subsistence forest use, this dependence might be a symptom of their poverty, and they may only break this cycle once they 'leave the forest' (Levang and Dounias, 2005; Chomitz, 2007).

Whilst needs for forest resources differ with a household's degree of wealth or poverty, there is great variation in who benefits from forests. Benefits often depend on factors beyond income and demographics and can be quite site specific (Cavendish, 2000; Suyanto et al., 2009). For instance, two villages bordering the same area of forest may use that forest in quite different ways (Jumbe and Angelsen, 2006; Dennis and Masozera, 2009) as might happen due to historical negotiation of customary access (Shaanker and Ganeshiah, 2004). The same applies within communities, where people are impacted by forest conservation interventions in diverse ways (Vira and Kontoleon, 2010).

The report "The Economics of Ecosystems and Biodiversity" (TEEB, 2010) also highlighted the disproportionate contribution of forests to the livelihoods of poor rural households, making up between 47 and 89 percent of the total sources of livelihoods for rural and forest-dwelling poor households ('the GDP of the poor'), while contributing much less to the overall national income, varying between 6 and 17 percent. Thus, forest-based livelihoods appear to be far more important to poor people as compared

to the national average in most countries, suggesting that it is these groups that are most vulnerable to the negative effects of forest loss. This becomes even more significant when additional considerations are taken into account, such as risk mitigation strategies (for example, forest product collection to cope with downturns in agricultural yields, and other income and consumption shocks) (Pattanayak and Sills 2001; Takasaki et al., 2004; Sunderlin et al., 2000; McSweeney, 2005). Avoiding deforestation and forest degradation may also have positive distributional implications in situations where the benefits from such actions flow disproportionately to poor households. On the other hand, if these benefits are predominantly captured by rich and powerful individuals, positive forest management outcomes may lead to increased inequality.



Returning from the buffer zone plantation with pine wood, animal fodder and bedding. Nyungwe National Park, Rwanda. Photo © Adrian Martin

This diversity in forest use patterns, and therefore the potential positive impacts of efforts to avoid forest loss and degradation, is shaped by complex political and social histories as well as cultural norms (Rugendyke and Son, 2005; Dressler, 2006; Malla, 2001; McLean and Straede, 2003; Himmelfarb, 2005). Added to this complexity, external drivers such as economic downturns (Sunderlin et al., 2001), changes in prices (Capistrano and Kiker, 1995) or policies (McShane et al., 2011) and attempts to improve infrastructure and technology (Angelsen and Kaimowitz, 1999) can have unforeseen and dynamic effects upon a household's well-being and its demand for natural resources at any particular point in time (Sheil and Liswanti, 2006). While addressing deforestation and forest degradation offers the potential to avoid the negative impacts of resource loss for forest-dependent people, the ways in which these strategies are governed and implemented can have important implications for the well-being of different social groups. The next section reflects on the experience of interventions across the agriculture-forest landscape, and the associated social and economic consequences of such interventions, especially for the most vulnerable and disadvantaged populations.

4.4 Social and economic consequences of management actions across the landscape

A landscape approach helps to capture the range of interdependencies and feedbacks between complex natural-human systems. As Chapter 3 (Section 3.2.) discusses, if REDD+ is to succeed, related interventions need to consider wider dynamics outside forests and consider the broader landscape (DeFries and Rosenzweig, 2010; Scherr and McNeely, 2007). The goals of a landscape approach are to reconcile demand-driven increases in agricultural production with protection and restoration of natural ecosystems, and maximisation of global and local ecosystem services in a socially-fair way. Spatial planning plays a crucial role within the landscape approach (Jackson et al., 2007; McNeely and Scherr, 2003), and strongly affects the distribution of economic and social costs and benefits across stakeholders in a landscape. An integrated landscape approach can better embrace both conservation and development objectives, and increase synergies among multiple local, regional and global societal objectives. Although reconciling competing objectives is never easy, such a holistic approach provides a platform for addressing diverse goals such as climate mitigation, food production, biodiversity conservation, the provision of ecosystem services, and delivery of economic, social and cultural development.

This section examines the social and economic aspects of four major categories of management actions across the landscape: actions aimed at addressing drivers of deforestation in the landscape, focusing on agricultural intensification (4.4.1); actions that occur primarily in forest areas, with a focus on protected areas (4.4.2) and sustainable use of forests (4.4.3); and actions aimed at restoring

or planting forests (4.4.4) that can take place across the entire landscape. These four management actions correspond to the major categories of direct intervention discussed in Chapter 3 of this report.

4.4.1 Addressing agricultural drivers

In tropical countries, the agricultural sector expands mainly at the cost of forest (Gibbs et al., 2010), while land degradation additionally depletes land potentially available for agriculture (Smith et al., 2010). Addressing drivers of deforestation and degradation within the agricultural sector is crucial for successful REDD+ interventions, as discussed in Chapter 3. Furthermore, conservation or restoration projects that ignore the immediate or underlying causes of forest loss are more prone to 'leakage' (Miles and Dickson, 2010), in which forest protection increases in one area, but as a result, deforestation increases elsewhere owing to demand for agricultural products. As practices are usually an outcome of deeply embedded social, economic and cultural histories, attempting to alter these drivers may be challenging as shown in Box 4.1.

The remainder of this section focuses primarily on the socio-economic consequences of agricultural intensification as an emerging priority in a world with a rapidly growing population.

Sustainable increases in the productivity of extensively-farmed agricultural lands (including abandoned and fallow areas) has been suggested as a key tool to reduce deforestation pressure and thereby release land for forest restoration (Beddington et al., 2012; Foresight, 2011; Godfray et al., 2010; Phalan et al., 2010). Agricultural intensification can be achieved through the adoption of multiple cropping, rotational systems, conservation agriculture and transition to *agroforestry*, among other

The use of fire in agriculture

Box
4.1

The use of fire in agriculture ('*slash-and-burn*') provides a good illustration of deeply-rooted agricultural practices with negative outcomes on forests. On the one hand, burning forests to clear land for agriculture results in a range of negative economic and social impacts (Barlow et al., 2012) such as respiratory illnesses (Kunii et al., 2000), losses of crops, livestock and farm infrastructure (Cochrane, 2003; Vera-Diaz et al., 2002), reduced availability and value of timber (Barlow and Peres, 2008; Gerwing, 2002; Barlow et al., 2010) and non-timber forest products (Nygren et al., 2006; Sinha and Brault, 2005). Reversing these drivers could therefore result in increased well-being for rural populations. On the other hand, farmers frequently favour fire use as it is a less labour-intensive method of production, and is culturally embedded within many societies (Barlow et al., 2012). As smallholders that practice slash-and-burn agriculture are among the poorest people in rural tropical forest landscapes (Hirsch et al., 2010), it is especially important that interventions take traditional knowledge and social and economic preferences into account (Parrotta and Trosper, 2012).

alternatives. In the Philippines, for example, improved small-scale irrigation systems in the lowlands increased labour demand and wages, attracting labour from a more extensive agricultural sector in the uplands. As a consequence, forest clearing was reduced by almost 50 percent (Shively and Pagiola, 2004). Converting low-productive pasturelands into silvopastoral systems can increase and diversify the output per unit of area, and may contribute to risk reduction, higher incomes and financial resilience while also reversing soil degradation (Murgueitio et al., 2011; German et al., 2006), increasing *carbon sequestration* and diminishing the need for further encroachment onto new lands (also see Chapter 3, Section 3.2.1).

The potential ‘*land-sparing*’ effect of increased productivity has gained increased attention recently, as part of a supposed (and arguably false) dichotomy between ‘*land-sharing*’ (Perfecto and Vandermeer, 2010) and ‘*land-sparing*’ (Phalan, et al., 2011). Several case studies (Lapola et al., 2010; García-Barrios et al., 2009; Tscharnke et al., 2012) suggest that increased productivity does not necessarily lead to land sparing (often quite the opposite) essentially because of the ‘*rebound-effect*’ (Lambin and Meyfroidt, 2011) - a classic economic effect where increased productivity makes an activity more attractive, leading to an increase in demand for its inputs (in this case, forestland). The rebound effect is particularly likely when the main crop being intensified is traded internationally, when the intensification occurs via labour-saving technologies and when increased profits can be used to clear new forestland for further expansion (Angelsen and Kaimowitz, 2001). When the dominant effect of increased productivity is a ‘rebound effect’, economic gains and losses are context specific. They will depend fundamentally on the balance of benefits from increased production (if any) and the losses in ecosystem services from increased deforestation.

However, increased productivity has the potential to lead to land-sparing when certain conditions are in place. These conditions include measures such as increased clarity of tenure and property rights, the equitable distribution of land rights, improved governance and better law enforcement, incentives for the sustainable management of forests, and recognition of the full range of ecosystem services that are associated with forests, thereby increasing the attractiveness of forests as a desired land-use, and reducing drivers for forest conversion.

Impacts of increased productivity on land owners are dependent on a series of variables. Most importantly are the tenure, governance and institutional conditions discussed in Section 4.2. If appropriate enabling conditions are in place, farmers who have decided to implement a new technique, lease their land and work for wages, or sell their lands, have usually done so in their own interest and are positively affected by the change. For example, the Foresight Project (Foresight, 2011) documented over 40 successful examples of sustainable intensification in agriculture. The results of these projects, gathered by early 2010, estimated that benefits reached 10.4 million farmers and their families, and improvements were apparent on approximately 12.75 million hectares of land.

Indeed, where there was domestic political, institutional and economic recognition that ‘agriculture matters’, agricultural outputs could be sustainably increased. Further, these examples demonstrated potential co-benefits, such as strengthening of environmental services, national domestic food budgets and improved local economic growth, the development of new social infrastructure and the emergence of new businesses.

Growth in smallholder agriculture has a disproportionately high impact on poverty reduction (compared to growth in other sectors), because smaller scale farmers are often subsistence farmers whereby agricultural outputs from the farm, mainly food and fodder, are directly used to sustain the family and on-farm livestock (de Janvry and Sadoulet, 2010; Loayza and Raddatz, 2010; Herrero et al., 2010; McDermott et al., 2010).

When enabling conditions are partially or completely absent, however, farmers might be forced to work against their will or be misinformed about their choices and corresponding rights and benefits. Detrimental effects of competition for land that drive the global rush for new agricultural land known as a ‘land grab’ have been demonstrated elsewhere (Zoomers, 2010; Lavers, 2012; Afionis, 2012; Borras and Franco, 2012). Land-grabs can also be especially problematic when land transfers occur under the guise of explicit environmental objectives - as may be the case under REDD+ - so-called ‘green grabs’ (Fairhead et al., 2012). While there may be positive aspects to facilitated international land acquisitions - including poverty alleviation, job creation or improvements in infrastructure - in practice, large-scale international land transactions are often accompanied by negative in-country effects, such as the loss of livelihoods and displacement of local populations (the poorest are usually the first to lose their land) which in turn leads to further degradation and deforestation (Zoomers, 2010).

When land-sparing results from increased productivity, other stakeholders benefit from increased provision of ecosystem services (due to avoided deforestation). These benefits tend to increase with the level of forest dependence. If overall agricultural output does not decline, and there is an increase in aggregate benefits from natural capital, the well-being of non-farming rural stakeholders is likely to increase. For stakeholders directly involved in the associated agricultural supply chain, positive regional distributive impacts might also occur.

Agricultural intensification might lead to job gains or losses depending on the labour-intensity of the new techniques. If job losses occur, re-training activities might be required to mitigate negative impacts, assuming there are suitable off-farm employment opportunities. Mechanisation might lead to a reduction in jobs in the rural sector but increased jobs in urban areas in sectors directly and indirectly related to the production of machinery. In this context, understanding winners and losers and the social aspects of adoption of more intensive systems is critical (Briske et al., 2011). Transition to more intensive systems also requires additional financial and labour investment by farmers, provision of training, extension, market support and marketing organisations.

4.4.2 Protected areas

Protected areas (PAs) are likely to be one of the preferred management approaches for implementation of REDD+ (as also discussed in Chapter 3). While their primary purpose is biodiversity conservation, many protected areas have multiple objectives, including socio-economic ones. In so far as protected areas help to conserve the ecosystem services provided by natural environments (Balmford et al., 2002), they have the potential to benefit people dependent on these services (Dudley et al., 2010). As discussed in Section 4.3., the loss of these services often disproportionately affects the poorest and most vulnerable groups, so these groups stand to benefit from forest protection. However, these groups can also suffer costs associated with protected areas, such as reduced access to land and other resources, increased exposure to harm from wild animals, gender inequities and loss of self-determination. Such costs can be particularly acutely felt where corresponding benefits are not fully realised, for example because they are captured by wealthy and powerful elites. The benefits and costs from protected area conservation therefore require careful consideration and management, an issue that this section explores in greater detail.

The imperative for protected areas as a tool for biodiversity conservation has remained strong and has led to rapid growth in publicly-managed forests (although other forms of governance of protected areas exist and are becoming increasingly prevalent – also see Chapter 3, Section 3.2.2), often including strict regulation for human access and use (Agrawal et al., 2008). In practice, the majority of protected areas are under sustainable use zones.

Despite limited knowledge about the rates of human use and occupation of protected areas, information which does exist indicates relatively high occupancy and use. Subsequently, there is a need to consider the social and economic impacts of PAs (Amend and Amend, 1995; Kothari et al., 1989; McNeely and Scherr, 2003). The tensions between protected areas and the livelihoods of local people in the tropics has been widely explored and contested (e.g. Wells and McShane, 2004; Curran et al., 2009; Miller et al., 2012). Whilst there are many case studies that present local evidence of benefits and costs of protection, there is concern about the consistency and rigour of available datasets (Ferraro and Pattanayak, 2006) and, in particular, social impact evaluations. Looking at aggregate national data, there seems to be substantial evidence that people living in and around terrestrial protected areas are poorer compared to national averages, but no evidence that the protected area is the cause of this poverty (Brandon and Wells, 2009; Ferraro et al., 2011; Chomitz, 2007). The fact that protected areas are designed to conserve biodiversity, which in turn underpins ecosystem services (see Chapter 2, Section 2.3), signifies that the potential for protected areas to support local livelihoods is significant, and there are many examples around the world whereby communities depend on goods and services derived from sustainable use strategies in protected areas (Dudley et al., 2010; Mansourian et al., 2008). Such aggregate data however, does not capture

local differences and there remains considerable concern about the unequal distribution of costs and benefits of protected areas, especially where the poorest shoulder the burden of costs (Adams et al., 2004), such as when people's homes or livelihoods are displaced to make way for conservation (West et al., 2006; Brockington et al., 2006).

Furthermore, aggregate economic analysis does not capture non-economic costs associated with conservation that are frequently important to human well-being. Indigenous and local people frequently make claims about conservation injustices that arise from loss of cultural recognition, right to self-determination, and the autonomy to live according to locally-conceived cosmologies of nature and society (Whiteman, 2009; Galloway-McLean, 2009). Thus, addressing social justice issues surrounding protected areas relates not only to distribution of economic costs and benefits, but also to political engagement in matters of culture and authority (Martin and Rutagarama, 2012).

Reflecting the fact that costs and benefits are still typically framed in largely economic terms, protected area trade-offs are often managed through forms of compensation or benefit sharing for local people (Wells and McShane, 2004). This can be effective at reducing conflict, although the relatively wealthy are often found to benefit more than those who are poorer, as shown for example in studies of revenues from gorilla tourism in Uganda (Sandbrook and Adams, 2012).

Overall, protected areas are generally considered effective at biodiversity conservation, but less effective at delivering socio-economic benefits (Naughton-Treves et al., 2005), and especially at distributing these equitably. Miller et al.'s (2012) review of papers finds nearly twice as many studies in which conservation adversely affects communities as those that find that conservation benefits communities. The resulting asymmetries between 'who benefits' and 'who loses' from protected areas are often framed as a concern for equity or justice, and as a trade-off between effectiveness, efficiency and equity. Where issues of social and economic equity are not addressed, there are uncertainties about how sustainable protected areas will prove in the long-term. As discussed in Section 4.5 below, decentralisation and community management of protected areas have the potential to improve both equity and sustainability, although local needs will not always be fully aligned with the long-term needs of biodiversity conservation.

4.4.3 Sustainable use in forested landscapes

This section explores the socio-economic implications associated with attempts to shift to more sustainable forms of the extraction of timber, wild animal resources and non-timber forest products (NTFPs).

Reduced impact logging

For large-scale timber extraction, reduced impact logging (RIL) has been promoted for its potential to maintain the carbon sequestration and other ecosystem functions of tropical forests (Putz et al., 2012; Fisher et al., 2011).



Medicinal plants harvested from forests contribute to traditional health care needs and can provide local livelihood benefits in rural communities. Village market in northern Madhya Pradesh, India. Photo © John A. Parrotta

Socio-economic benefits include a reduction in personal hazards suffered by forest workers and the ecosystem services provided to forest-dependent communities relative to conventional logging (Putz et al., 2008). However, others suggest that RIL operations are insufficient to maintain the levels of environmental services provided by tropical forests in their pre-logged state (Boltz et al., 2003). Furthermore, RIL faces a number of obstacles to widespread adoption across diverse geographical and institutional scenarios (Boltz et al., 2001). In some contexts, RIL presents clear financial disincentives as the opportunity costs of setting aside timber stands to maintain ecosystem integrity can be substantial (Tay, 1999). Dramatic changes in market signals – for example, increased premiums on certified timber or the generation of carbon credits – may be needed before conventional forestry operations adopt RIL (Boltz et al., 2003). Additionally, information dissemination on the benefits of RIL, as well as technical assistance, training and pilot programmes may contribute to increasing RIL uptake (Bacha and Rodriguez, 2007).

In some contexts, community-based timber extraction through cooperatively-managed community enterprises or community-private joint ventures has enabled smaller scale harvesting with local benefits (FAO, 2006; MacQueen, 2008; Oberndorf et al., 2007). The success of such ventures in reaching the poorest households has, however, been mixed as people often lack the formal rights necessary to benefit from timber. The political economy of timber production tends to favour large scale, politically-connected operators (Belcher, 2005), while local associations take time to develop market knowledge, linkages and governance capacity (MacQueen, 2008;

Donovan et al., 2008). Furthermore, national and sub-national regulations can constrain small enterprises in unfair ways, making the development of supportive institutions a pre-requisite for the effective and equitable operation of community-based timber enterprises (Warner, 2007; Dunning, 2007; Stoian, 2005).

Non-timber forest products

As discussed in Section 4.3, the significance of NTFPs for subsistence, income, farm inputs and reduced vulnerability, especially for poor households in forest landscapes, has been well documented in diverse contexts (Marshall and Newton, 2003; Stoian, 2005; Belcher, 2005; Fisher et al., 2008). However, sustainable NTFP harvesting may only offer limited scope to shift forest dwellers out of poverty where there are weak or informal local rights to the resource, institutional, capital and technological barriers to effective marketing, skewed value chains, and uncertain niche markets (Belcher, 2005, Belcher and Schreckenberg, 2007). These same factors can also contribute to NTFP overharvest in environments where harvesters do not enjoy a secure stake in the ongoing management of the resource (Belcher, 2005). Sustainable governance of NTFP use therefore needs to be founded upon an understanding of NTFP value chains (Belcher and Schreckenberg 2007; Stoian 2005) and the place of NTFPs in livelihoods and landscape use (Laird et al., 2010; also see Chapter 3, Section 3.2.3.2).

Bushmeat

Like NTFPs, the sustainable hunting of wild species (bushmeat) can have major livelihood significance for the poor, whether for direct use (Brown and Williams, 2003) or for income (Coad et al., 2010; de Merode et al., 2004). The lack of recognised rights is a major challenge in bringing wild trade into more sustainable management, with bushmeat harvest considered illegal in many contexts (Brown and Williams, 2003). Given the livelihood costs of exclusion, outright restrictions rarely succeed in preventing hunting (Adams and Hulme, 2001). Instead, current thinking emphasises the delineation of local rights as well as systems of regulation, certification and/or chain of custody to monitor harvest and trade (Brown and Williams, 2003; Laird et al., 2010; Nasi et al., 2008). For both NTFPs and bushmeat, community-based systems of management may have a role in sustainable use. However, community management of extractive use may not work where there is an insufficient wildlife resource, where markets have unsustainable levels of demand, and where the use restrictions in sustainable management do not match its economic benefits (Adams and Hulme, 2001; also see Chapter 3, Section 3.2.3.3).

In many landscapes, concurrent extractive use may need to be managed; for instance, timber, NTFPs, hunting, *shifting cultivation* and/or agro-forestry (Garcia-Fernandez et al., 2008). Latin American research highlights that trade-offs are often involved between timber and NTFP extraction (Garcia-Fernandez et al., 2008; Guariguata et al., 2008). Timber, with its higher value, tends to ‘trump’ NTFPs as a priority in multiple forest management

without supportive and integrated institutions that foster diverse uses (Garcia-Fernandez et al., 2008). Other significant factors that can help to balance multiple use include effective devolution of authority and local rights; appropriate technical and institutional capacity; economic viability and distribution of revenues; reconciliation between local, national and global interests; and the development of effective, trusting relationships to underpin long-term, flexible approaches (Radachowsky et al., 2012).

4.4.4 Restoration, reforestation and afforestation

Restoration, reforestation and afforestation have been proposed under REDD+ for the enhancement of carbon stocks. Such actions range from active planting to passive natural regeneration, with the results varying in ecological complexity from *monoculture plantations* to diverse *secondary forest*. Depending on a number of factors, such as overall landscape condition or prior land use (see Chapter 3, Section 3.2.4), all of these actions can provide some degree of biodiversity and carbon benefits. The relationship between social impacts and either biodiversity or carbon impacts in restored or *planted forests* is much more equivocal. In fact while large areas of land are technically suitable for reforestation, afforestation or restoration, it has been estimated that only about a third to a tenth of that is available once social and political considerations are taken into account (Bass et al., 2000).

While there is substantial literature on the impacts of *plantations* on local communities, less has been written about direct and measurable social impacts of ecological restoration (Bullock et al., 2011; Birch et al., 2010). Indeed plantations, particularly those of fast growing *introduced species*, have been criticised over the last decades for their frequently negative impacts on local communities (Charnley, 2005; Gerber, 2011; Barr and Sayer, 2012). As a result, many companies have been improving their practices, and certification schemes (notably the Forest Stewardship Council) are also paying more attention to such issues within plantations. Ecological restoration efforts on the other hand, have increasingly been perceived as having the potential to offer higher local benefits, if certain pre-conditions, such as clear land rights, are in place (Fox et al., 2011).

Planted forests, as defined by the FAO, can be both for productive and for protective purposes (FAO, 2010). On the other hand restoration, through a diversity of means (*assisted natural regeneration*, passive natural regeneration, enrichment planting etc.), tends to be associated with ecological objectives (Lamb et al., 2005; Alexander et al., 2011). Approaches such as *forest landscape restoration* aim to balance both human well-being and ecological priorities within landscapes rather than at the site level (Mansourian et al., 2005a). The ultimate objective (timber or pulpwood production, watershed or coastal protection, biodiversity conservation, fuelwood provision, carbon sequestration etc.) of the restoration, reforestation or afforestation effort will have repercussions on the amount of funds required, the type and intensity of

management applied, and the degree of local empowerment, all of which will impact on local welfare.

Local socio-economic benefits from restored and planted forests

Socio-economic benefits from restored or planted forests can be divided into three levels: increased local opportunities, improved financial and other incentives, and enhanced availability of ecosystem goods and services. Local people may benefit through increased opportunities from, for example: greater access to markets through the creation of new roads by plantation companies, the provision of various services (roads, water, electricity) associated with plantation companies and jobs. An increase in local investment (by the government and/or plantation companies) can contribute to overall economic development for the region with subsequent benefits to local communities. Local communities may also become politically empowered as they become active participants in restoration schemes. In many cases, in order for restoration to take place, the clarification of land rights proves necessary and therefore, the restoration process provides an avenue for improvements in governance and property regimes (Oviedo, 2005; Mansourian et al., 2005b; Rands et al., 2010).

At a second level, financial or other incentives may be provided for engaging in restoration or plantation activities, including payments in cash or kind, as well as employment benefits associated with collecting seeds, tending tree nurseries or planting trees (Nawir et al., 2003; Bass et al., 2000; Pokorny et al., 2010). For example under China's Grain for Green programme established in 2002, farmers receive grain for planting trees on degraded slopes (Fox et al., 2011).

At a third level, restoration may provide a range of ecosystem goods and services upon which local communities depend. These include timber, firewood, food, medicinal plants and other NTFPs (Jindal et al., 2008; Salafsky and Wollenberg, 2000; MA, 2005), and may also include intangible values that are of cultural or spiritual significance. Restored forests can also supply protective benefits such as, for example, coastal protection, soil stabilisation or water filtration (MA, 2005).

Table 4.2 below provides examples of projects and policies aimed at restoring forests for the provision of a range of ecosystem services.

Decisions concerning plantations or restoration frequently fail to take into account the socio-economic value of restored or planted forests which extends well beyond a store for carbon and the provision of timber, and includes a vast range of ecosystem goods and services, some with relatively easily quantifiable values (for e.g., nuts or fruit) others much less so (for e.g., spiritual values) (TEEB, 2009; Birch et al., 2010; Bullock et al., 2011).

Local socio-economic costs of planted forests

The most significant costs associated with planted forests can be categorised as opportunity costs, loss of control and decision-making power, loss of access (to land, forests, forest products), and environmental health concerns.

Restoration of ecosystem services		Table 4.2
Ecosystem service restored	Example	Social/economic outcomes
Soil stabilisation	The Capital Development Authority in Tanzania's capital, Dodoma, has established a forest belt around the city (Chamshama and Nwonwu, 2004).	Soil and wind erosion has been controlled due to restoration of miombo woodland, and a space has been created for recreation.
Provision of food	In 2008, seven pilot centres under a World Agroforestry Centre project in Cameroon produced over 122,500 plants of indigenous fruit and nut trees (both for home consumption and for sale) (Asaah et al., 2011).	Over 10,000 farmers from 200 communities are benefitting in the agroforestry network in north and north west Cameroon.
Provision of medicines	Large scale reforestation programmes have been introduced under Vietnam's official "5 million hectare reforestation programme" (Dung et al., 2002).	Households in Bac Ha District in Lao Cai grow the medicinal plant <i>Amomum aromaticum</i> , harvesting 200-300 kg per year, worth 10-20 times more than rice cultivated on the same area.
Energy provision	In the Central Highlands of Kenya, trees have been planted on coffee and tea plantations (Chamshama and Nwonwu, 2004).	70-80 percent of households utilise firewood grown on coffee and tea farms.
Protection against natural hazards	Mangroves and <i>Casuarina</i> plantations were established along the Asian coastline (Danielsen et al., 2005).	In Cuddalore district in Tamil Nadu (India) five villages located within coastal <i>Casuarina</i> plantations experienced only limited damage from the devastating 2004 tsunami.
Improved agriculture and agroforestry	The Fandriana Marolambo Forest Landscape Restoration project has been implemented by WWF since 2004 to conserve, sustainably use and restore rainforests in Madagascar (Roelens et al., 2010).	A total of 8,400 inhabitants (1,400 families) have directly benefited from improved rice production, crop fertilisation and diversification.
Cultural values	Native species restoration programmes have been introduced in New Caledonia's unique dry forest (Mansourian and Vallauri, 2012).	Local pride in indigenous species has been restored, as shown by a significant increase in demand for native tree species (reflected by their increased prevalence in tree nurseries).
Knowledge about indigenous species and restoration	Between 2007 and 2008 IFAD led a forest restoration programme in Cameroon with local farmers (Asaah et al., 2011).	Improved skills and knowledge about local trees, and increase in household tree planting.
Education and awareness raising purposes	Tree nurseries and restoration areas have been established under UNEP's "one billion trees campaign" and the late Nobel prize winner Wangari Maathai's "Green Belts Movement" (Mansourian et al., 2005a).	Provision of a tangible and readily understood conservation message, and promotion of education and awareness about restoration.

These costs are typically spatially unevenly distributed with many plantations having been established to provide benefits to non-residents while replacing land uses that were providing benefits to local communities (Jindal et al., 2008; Oviedo, 2005; Paoli et al., 2010; Mayers and Vermeulen, 2002). For instance, in Lao PDR the rapid expansion of rubber plantations is replacing agriculture and affecting food security of rural communities (Fox et al., 2011).

A recent meta-analysis of plantations identified land tenure as the main source of controversy surrounding plantation establishment (Gerber, 2011; Wilson, 2009) which has plunged in some cases already vulnerable populations further into precarious living conditions. While jobs may be generated by plantation companies, they are frequently for skilled, seasonal and migrant workers

rather than for local communities (Bass et al., 2000; Pokorny et al., 2010; Nawir et al., 2003).

Plantations may also drive up the price of land (Carriere and Lohmann, 1996) as they may stimulate demand for land, thus creating extra financial stress to local people forcing them to move to urban areas. This in turn may lead to a general destruction of the social fabric and ultimately, to conflict (Barr and Sayer, 2012). Lack of trust, insensitivity to local culture, mis-communication and limited technology transfer, have also been associated with plantation establishment (Nawir et al., 2003; Fox et al., 2011). A further source of mistrust arises from the significant benefits that have been granted to plantation companies around the world (Barr and Sayer, 2012). Such incentives include provision of free inputs, grants and loans, subsidies, tax concessions, joint venture

Socio-economic and ecological impacts of forest restoration, reforestation and afforestation – illustrating synergies

Fencing areas of degraded forest to promote natural regeneration

Historical clearing of miombo woodlands in the Shinyanga region of Tanzania was stimulated by a desire to eradicate tsetse flies, the expansion of cashcrops, fuelwood plantations and general policy failures. In 1985 the Sukuma people living in Shinyanga decided to revive traditional methods of restoration, which promoted natural regeneration through fencing off certain areas in order to protect them from cattle. Fifteen years later, the area concerned had been dramatically transformed with tree cover returning. In this case communities benefited (more crops, better fodder and fuelwood), carbon sequestration was increased and the landscape was enhanced for biodiversity (Barrow et al., 2002).

Planting trees around a protected area to create a buffer zone

Community forestry has been encouraged around Nepal's Chitwan National Park as a means of reducing pressure on the park and to provide fuelwood and other products for local communities. Tree plantations were established in severely degraded areas, and natural regeneration was promoted in less degraded forest habitats. A perverse result of forest restoration in this buffer zone was an increase in human–tiger conflict as tigers were able to roam beyond the limits of the park (Gurung et al., 2006). In this case there are both significant costs and benefits to local communities while for biodiversity and carbon, benefits can be considered positive.

Planting mixed indigenous species with the assistance of local people to produce locally useful species

In Madagascar a WWF project initiated in 2003 in the moist forest landscape of Fandriana-Marolambo engaged local communities in the collection and management of seedlings with the aim to restore forest goods and services. While in 2007 communities planted only introduced species, by 2010 of the 328,416 seedlings planted, over 80 percent were local. This result was thanks to significant efforts by local facilitators and teams to engage with the communities, train them and better define together their needs in terms of agroforestry and overall landscape condition (Roelens et al., 2010). In this case communities benefited from improved knowledge and a greater diversity of crops, carbon sequestration was increased and biodiversity is likely to benefit in the long term with an increase in natural forest cover and a reduction in fire and in plantations of introduced species.

arrangements, the creation of an enabling environment and removal of structural impediments (Enters et al., 2004; Evans and Turnbull, 2004; Bohm, 2008). This is seen as unfair preferential treatment for large companies that extract profits from the local landscape without necessarily providing adequate benefits to the surrounding communities.

Several negative environmental health impacts have also been attributed to plantations, particularly fast-growing plantations of introduced species with the primary purpose of producing pulp and paper. For example, silvicultural techniques may affect soils, air and water quality, all of which may negatively impact the health of local communities (Jindal et al., 2008; Evans and Turnbull, 2004; Schirmer and Tonts, 2002).

Local socio-economic costs of restoration

Forest restoration also represents a potential opportunity cost to local communities as land that is set aside to be restored could have been used by local communities for alternative purposes. The financial cost involved in reforestation can be high compared to alternative options (passive restoration, removal of barriers to restoration etc.) for enhancing carbon stocks, particularly if these costs are not covered by the sale of carbon credits for example or through the sale of timber and NTFPs (Van Kooten et al., 2004; Birch et al., 2010). The extent of the opportunity cost will depend on the degree of local dependence on ecosystem goods and services, and on the availability of alternative options. For example, if restoration implies fencing off certain areas to enable natural regeneration, local herders may find their potential grazing land

severely reduced. Thus, the weighting of costs and benefits across different stakeholder groups also needs to be considered.

Box 4.2 discusses three case studies which illustrate the complexity of interactions between carbon enhancement, biodiversity and people in the context of planted and restored forests, and how synergies may emerge between these different objectives.

The political and economic context within which restoration or plantation forestry takes place determines the flow of benefits (or costs) to the poorest, most vulnerable communities. While it is possible to achieve gains in biodiversity conservation, carbon sequestration and human well-being, appropriate enabling conditions need to be in place (governance, institutions and tenure) for all three to occur, especially if the flow of benefits to the poorest and most vulnerable is to be prioritised.

4.5 Major approaches for implementing management actions

The way in which management actions are governed will determine the strengths and weaknesses of implementation, including the capacity to support rules and institutions related to resource use and benefit sharing. State-controlled governance has historically been the norm for large-scale conservation interventions (Brockington, 2002; Adams, 2004). In principle, this involves the application of scientific knowledge to state planning systems, supported by forms of top-down regulation justified by the need to protect the public good. In the last few decades, this top-down regulatory approach to governance

has given way in some places to more networked, partnership approaches to governance of natural resources which extend the range of legitimate knowledge and stakeholders (Adams and Hulme, 2001; Pretty, 2002). These more participatory and decentralised forms of governance can also gradually shift the ways in which rules are enforced, away from hierarchical sanctions to institutional norms based on trust and reciprocity (Ostrom, 1990; Baland and Platteau, 1996). More recently, a third distinct governance approach has become popular, based on the extension of market mechanisms to environmental conservation (Daily and Ellison, 2002; Balmford and Whitten, 2003). Sometimes referred to as ‘neoliberal’ (see, for instance, Arsel and Buscher, 2012), market-based governance relies on the development of markets for ecosystem services enabling social preferences for sustainability to be expressed through market transactions, regulated by the decisions of individuals and businesses to enter and exit particular markets. These approaches are not as mutually exclusive as they might at first appear and it is not unusual to find all three interacting in the same location (see Box 4.3.). Both decentralisation and market-based approaches are currently playing important roles in REDD+ pilot projects and are therefore considered more critically in the next section.

4.5.1 Forest decentralisation

Despite a shift towards greater community- and privately-managed tropical forests, the majority of tropical forests remain formally under public ownership (RRI, 2009). There is also considerable regional variation, with 98 percent of forests still publicly administered in tropical

Africa (RRI, 2009). This (patchy) trend towards diversity of tenure and authority, can be distilled into two overarching lessons of relevance to REDD+. Firstly, decentralised, community forestry (in its myriad forms) can be effective in terms of ecological outcomes; and second, the distribution of authority and resources is important to both effectiveness and equity.

Large scale studies do not support the view that either private, public or community forest ownership is automatically more effective for conservation (Agrawal et al., 2008). This is a significant shift from Hardin’s (1968) influential thesis that only private and public ownership could achieve adequate management of the ‘commons’. Whilst there is widespread appreciation of the difficulties of community forestry, it is equally recognised that it has the potential to resolve some of the problems associated with weak state governance of forests, namely failures of enforcement, benefit sharing and livelihood protection (Agrawal and Angelson, 2009). Empirical evidence for the effectiveness of community management has been mixed, but has become far more robust as it moves beyond case study work. Persha et al.’s (2011) study of 84 sites found that conservation and livelihood synergies are more likely where there is local participation in governance. Ostrom and Nagendra (2006) and Nepstad et al. (2006) find evidence that community and indigenous management can protect forests in Amazonia whilst Bray et al. (2008) also find livelihood benefits in this region. Porter-Bolland et al.’s (2011) study of 40 PAs and 33 community-managed forests found that community-managed sites had lower and less variable deforestation rates whilst Hellebrandt et al.’s (forthcoming) review of 21 studies

Box 4.3

Participatory Forest Management and REDD+ in Tanzania

Tanzania’s approach to Participatory Forest Management (PFM) involves devolution of forest management responsibilities to village councils and assemblies. Participatory forest management includes joint forest management (co-management) of state reserved forests and community-based forest management of village lands. By 2008, 11 percent of Tanzania’s forest land was under PFM arrangements, involving 18 percent of Tanzanian villages (Blomley et al., 2008). Between 1997 and 2007 basal areas and volumes for those forests under PFM had been found to increase, whereas those under state management experienced declines. Incidents of illicit activities also declined in forests under PFM and, compared to non-joint management, incidence of fire declined by a factor of six (Blomley et al., 2008). Income and livelihood impacts remain less certain, with relatively low income from forest management and evidence that a larger share of benefits are captured by elites (Meshack et al., 2006). Where transaction costs are high relative to benefits, there appears to be no obvious reason for local communities to sustain PFM. The explanation for them doing so might be that it is in the interest of the elite minority who ensure community support is sustained (Blomley et al., 2008). The paucity of benefits, and weak governance of benefits, pose a critical problem for the multiplication of PFM and its sustainability (Blomley and Ramadhani, 2006; Burgess et al., 2010).

REDD+ piloting in Tanzania has been integrated with PFM. As such, it provides some potential to respond to PFM’s income gap, but also poses some risks. For example, the Mpingo Conservation and Development Initiative has used REDD+ as a way to increase communities’ revenues from forest management and to move away from purely donor-funded PFM expansion. They secured Tanzania’s first Forest Stewardship Council (FSC) group certificate in 2010 and aim to make PFM self-sustainable through community sales of commercially valuable timbers such as *Dalbergia melanoxylon*. The estimated average income of USD 14 ha⁻¹ yr⁻¹ from timber sales exceeds the anticipated gross income from carbon sales (an expected 1.8tC02e ha⁻¹ yr⁻¹ at forecast USD 5 market price). However, the difficulty with expanding the PFM/FSC model is that forests are typically degraded and restoration is required in a first instance, thus significantly delaying the flow of income.

Whilst it is too early to evaluate this and other REDD+ interventions, it is clear that the donor and NGO project model of REDD+ is leading to innovative practice involving new income streams to support community forestry. The opportunities for communities are significant, but governance issues will be critical to realising these benefits.

provides some evidence that community management is more likely to support poverty alleviation.

The conditions that determine whether community forestry succeeds are likely to provide important lessons for REDD+. These conditions are far from fully understood, and are clearly complex, but some well-established lessons that will apply to REDD+ can be identified. Several case-based and large-scale studies have confirmed that the level of devolution of authority is critical and often undermines forest decentralisation (Ribot et al., 2006; Sikor and Thanh, 2007). Local rule-making autonomy is associated with better environmental and livelihood outcomes (Chhatre and Agrawal, 2009; Hayes, 2006; Hayes and Persha, 2010). However, this is only the case if decision-making authority is genuinely transferred to the local level, and related institutions foster downward accountability (Ribot et al., 2006). Importantly, the communities that gain management rights over resources are neither homogenous nor equitable by nature (Agrawal and Gibson, 1999; Agarwal, 2001); therefore equitable outcomes will only occur where rules exist around procedural and distributive equity (McDermott and Schreckenberg, 2009). Participation, especially in a gendered context, needs to be both symbolic and substantive to improve forest governance and enhance social equity (Agrawal, 2010). Some of these issues are illustrated in Box 4.3, which examines the experiences of participatory forest management and REDD+ in Tanzania.

4.5.2 Payments for ecosystem services

Payments for ecosystem services (PES) refer to voluntary contracts that enable a buyer to purchase a defined ecosystem service (or a land-use likely to secure that service) from the 'provider', on the condition that the provider carries out agreed actions to sustain provision of that service (Wunder, 2005). Originally promoted as an efficient and effective market-based approach to redressing environmental degradation, improvement in social welfare was not an explicit objective of PES (Wunder, 2005). However, the growing uptake of PES to support biodiversity and watershed protection, as well as forest carbon sequestration and storage, has raised concerns on whether the financial benefits of PES outweigh the livelihood risks for rural resource users and managers where PES involves restricted access to forest resources and associated changes in subsistence strategies. The early evidence on livelihood risks and benefits, which is strongest from Latin America (Alix-Garcia et al., 2005; Grieg-Gran et al., 2005; Blackman and Woodward, 2010) compared with the more recently established schemes of Asia and Africa, reveals mixed livelihood outcomes that are strongly shaped by the institutional arrangements governing specific schemes as well as broader political and economic conditions.

Payment for ecosystem services schemes interact with the full gamut of financial, natural, social, human and built assets that underpin local livelihoods (Chambers and Conway, 1992; Landell-Mills and Porras, 2002; Tacconi et al., 2010), and most PES livelihood impact studies

consider impacts across these domains. While the income associated with PES schemes is typically small (Corbera, 2010; Wunder, 2008b; Leimona et al., 2009; Tacconi et al., 2010), payments have nevertheless been greatly valued by landowners in some cases (Rios and Pagiola, 2010). Some scholars argue that financial incentives alone are not the main pathway to either environmental benefits or livelihood improvements, and that attention to other assets, for instance to human capital through capacity building, and strengthened access to natural capital through more secure resource rights, will make a greater difference to overall livelihood outcomes (Leimona et al., 2009; Tacconi et al., 2010; Evans et al., 2012).

Regardless of the nature and scale of benefits, the ability to access any benefits ultimately depends upon the ability to participate in PES contracts. Several factors can hinder open participation in PES agreements, including entry requirements about size of land holdings and tenure, which exclude the rural landless (Porras et al., 2008; Larson, 2011), scheme location (Pagiola et al., 2008; Wunder, 2008a); and whether households have the human and financial wherewithal to negotiate agreements and implement the associated measures, for instance tree planting or allocating labour for forest patrols (Pagiola et al., 2008; Wunder, 2008a). In this way, broader political, economic and institutional processes, such as land tenure, existing imbalances in assets/wealth, and power differentials between buyers, sellers and intermediaries (Vatn, 2010; Thuy et al., 2010) ensure socially differentiated capacities to engage in PES schemes and to access any associated benefits (To et al., 2012). Indeed, governance failings such as weak or uncertain tenure can be a source of forest degradation and loss, and thus undermine the environmental outcomes of PES as well as contributing to inequitable livelihood impacts from PES schemes (McElwee, 2011). PES schemes must therefore be complemented with strategies to address these tenure and governance issues.

The design of PES agreements strongly determines the scope and distribution of impacts. The timeframe of schemes (which may be anywhere between seven and 100 years (Tacconi et al 2010), but most often in the 10-15 year range (Huang and Upadhyaya 2007)) and the scheduling of payments over the duration of agreements determine, for instance, how costs and benefits are borne out between generations of local resource users. The choice to make individual payments, collective payments or a combination of these, determines whether benefits accrue primarily to individuals, households or collective entities through investment in infrastructure and services (Mahanty et al., 2012). The distribution of benefits along value chains is another key concern; intermediary organisations have been known to receive as much as 40 percent of PES income to support their costs in facilitating PES transactions (Mahanty et al., 2012; German et al., 2010). This leads some to suggest that the inherent asymmetries in global markets, including those for ecosystem services, may ultimately be skewed against individual farmers or local collective entities (e.g. indigenous bodies) operating as ecosystem service sellers (Corbera and Brown, 2010; McAfee and Shapiro, 2010).

Institutions and political economy shape the benefits and risks of PES in Vietnam

In the twentieth century Vietnam's forests were nationalised and placed under the management of State Forest Enterprises. Customary claims to these lands were overridden in the 1990s and 2000s, and long-term (tradeable) use/management leases to degraded or barren 'forest' were issued instead (McElwee, 2011). As a result the country's forests remain unevenly titled and distributed, with conflict resulting from current and prior land claims (McElwee, 2011).

Since 2005, these forests have been the focus of several PES schemes which address watershed services, biodiversity and, more recently, forest carbon (McElwee, 2011). McElwee (2011) finds a mixed picture of participation in PES schemes, usually driven from the top down, where rights-holders with red book certificates are able to enter PES agreements, while those without formally recognised rights receive 'forest protection payments' – a lower amount only for their labour as their right to resources is unrecognised. In this way, PES has exacerbated and perpetuated initial disparities in land holdings (To et al., 2012). In some cases, those with unrecognised customary tenure have been excluded altogether from benefits (To et al., 2012) with subsequent conflict and sabotage of forest protection activities. In most cases, PES payments have tended not to reach the poorest households which lack the formal rights required to enter into PES agreements (To et al., 2012)

The evidence thus points to several trade-offs in the design and implementation of PES schemes. Designing PES schemes to be economically efficient and environmentally effective may diminish the focus on equity and social welfare (Wunder, 2008b; Pascual et al., 2010), though more research is needed to understand this relationship. Trade-offs between social equity and other PES objectives may be most striking where resource rights are unrecognised or ambiguous, the bargaining power of local resource users is weak (Scherr et al., 2004; To et al., 2012; Pascual et al., 2010) and collective agreements override consent at the individual and household levels (Milne and Adams, 2012). Some of these issues are illustrated in Box 4.4 which draws on the experience from PES implementation in Vietnam.

4.5.3 Certification and standards

Schemes for certifying the sustainable management of forests started formalising in the early 1990s, once it became apparent that a convention specifically related to forests was unlikely to be agreed in the near future (also see Chapter 5 for a discussion of global forest governance debates). These market-based schemes were prompted by the desire to ensure that environmental and social standards were being respected in the management of forests (Elliott, 2000). While the umbrella Forest Stewardship Council (FSC) was one of the first organisations to promote principles and criteria for the sound management of forests, there exist today over 50 certification schemes worldwide; with the majority falling under one of the two umbrella organisations, i.e. the FSC and the Programme for the Endorsement of Forest Certification (PEFC). Some forest certification schemes are systems-based (ie: defining environmental management systems and then ensuring that the forest organisation respects these) while others are performance based (i.e., defining levels of achievement and then assessing whether these levels are met by the forestry operations) such as the FSC (Elliott, 2000). The forest certification process covers four areas: the development of agreed standards defining sustainable forest management (these can be at the management unit, national or international level);

auditing of the actual forest operations and issuance of certificates to companies that meet those standards; auditing of the chain-of-custody to ensure that a company's products come from certified forests; and the use of labels on products that enable consumers to recognise certified products (FAO, 2011). As of 2010 the area of certified forests covered by the two main organisations (FSC and PEFC) totalled about 350 million hectares (FAO, 2011). By August 2012 the area under FSC certification alone totalled 162,328,116 ha in 80 countries (FSC website).

In principle forest certification is intended to be a market-based tool that secures better environmental and social practices. In practice, criticism has been levelled at certification schemes for their complexity thus potentially excluding the majority of small players. Such schemes have been viewed by many developing countries as a barrier to trade whereby those unable to afford certification would be excluded from the markets of developed countries. The complexity and significant costs involved with certification have proven to be an obstacle for communities or forest owners from developing countries, particularly as they are generally unable to reap the benefits of the price premium which frequently goes to middlemen (Thorner, 2003). The diversity of forest types, stakeholder groups and management approaches also signify that it may be difficult to apply the same principles and criteria across regions (Rametsteiner and Simula, 2003). Moreover, evidence shows that product commercialisation is generally associated with male dominance in value chains, while greater support to informal markets is more likely to improve benefits to women (Mwangi and Mai, 2011).

Nevertheless, in many instances the multistakeholder process involved with certification has enabled the voice of communities to be heard alongside that of powerful industry players (Hauffer, 2003). In some cases communities have benefited from improved tenure rights thanks to the certification process, for example in Guatemala and Bolivia (Molnar, 2003). Certified operations also provide better working conditions (Pokorny et al., 2010). Under the FSC principles, indigenous peoples' rights and knowledge are to be recognised, and their free prior and informed consent (FPIC) is required (principle 3 and

its respective criteria). The rights of forest workers and communities are further protected under FSC principle 4 (concerning health, safety, work opportunities, labour conditions etc.) and appropriate consultation among local stakeholders is required. Strengthening and diversifying the local economy is part of FSC's principle 5 related to the benefits of forest management.

It has been suggested that certification standards could support REDD+ by strengthening the social and environmental safeguards (Merger et al., 2011). The lessons emerging from forest certification schemes would therefore, be of benefit to inform REDD+. However, to date, there is limited data available on the impacts of forest certification, although anecdotal and case by case evidence suggests that certification schemes have in general provided a step in the right direction for forest management (Nussbaum and Simula, 2004). Three conclusions emerge from Nussbaum and Simula's (2004) review of certification: 1. different certification systems appear to address different needs of different users, 2. It is unclear to what extent impacts are scheme-specific or generic to certification, and 3. concerns remain about the impacts and equity of forest certification on different stakeholder groups (for example, small or community enterprises) (McDermott, in press). Paradoxically, currently the majority of certified forests lie in Europe and North America where both social and environmental concerns may be of less concern than in tropical and sub-tropical countries.

Forest certification has evolved markedly since the launch of the first schemes in the 1990s. While it remains a predominantly northern instrument targeting discerning consumers who have significant buying power, certification can be considered as a step in the right direction for mitigating the social and environmental impacts of the forest industry.

4.6 Lessons for REDD+ from previous policy and management approaches at the landscape level

Even though the precise form and governance of REDD+ is still evolving, the rich evidence base from prior forestry and agriculture interventions confirms that REDD+ will have social and economic impacts. This section discusses a range of lessons learned from the evidence discussed in the previous sections that could be useful to anticipate and address possible social and economic impacts from REDD+ interventions.

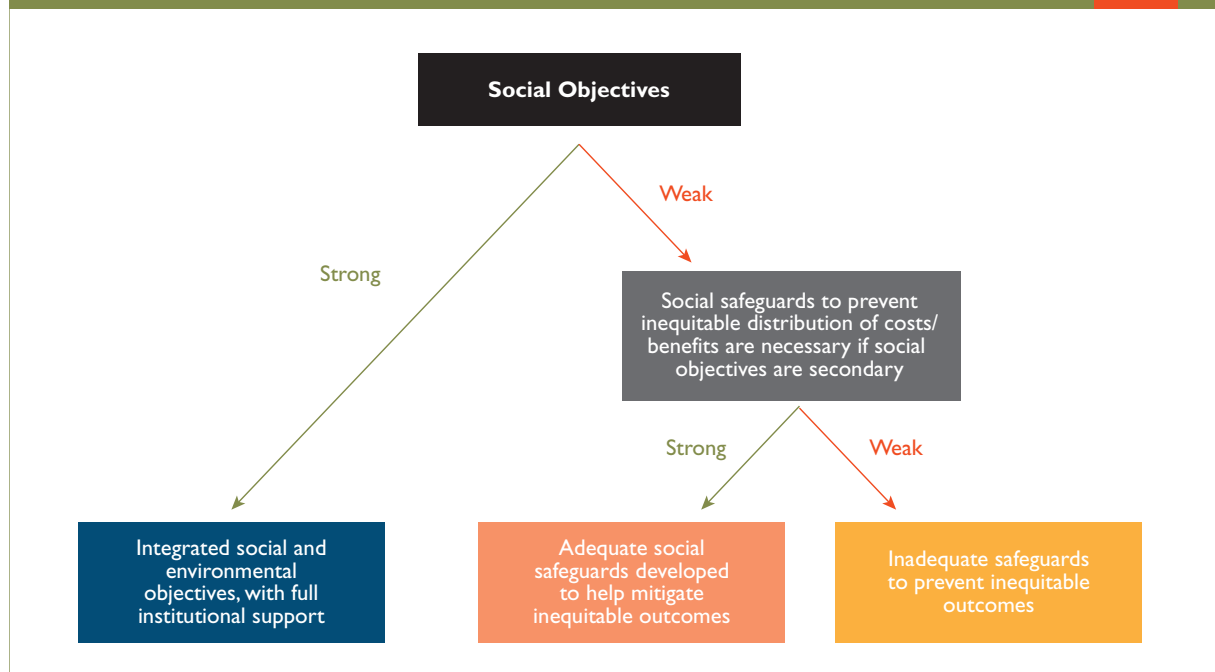
REDD+ presents an opportunity for the poorest individuals and groups in rural areas. Firstly, REDD+ interventions will aim to conserve or enhance the ecosystem services upon which the poor are most dependent. Secondly, REDD+ promises to leverage new sources of finance that can potentially reward rural people for environmental management.

However, these opportunities can easily be lost: the poor can be excluded as beneficiaries, for example as has been the case in many PES schemes with barriers to entry, or they can *de facto* be excluded as has often been the case in forest certification because of the high costs involved. REDD+ activities might even harm the poor where, for example, exclusion from benefits is accompanied by increased costs that arise from reduced access to resources.

Initiatives to enhance carbon and biodiversity that embrace and adopt clear social objectives are likely to minimise trade-offs between environmental objectives and livelihoods (Sikor et al., 2010; Sunderlin et al., 2009; Larson, 2011). The adoption of safeguards is a further way in which negative impacts on vulnerable stakeholders can be avoided, but does not necessitate as high a level of commitment to social and economic objectives. Figure 4.1 illustrates these two mechanisms through which social

Social and economic concerns in REDD+ and biodiversity interventions

Figure 4.1



and economic objectives can be reflected in REDD+ and biodiversity interventions.

Previous forestry-related interventions highlight the diverse and complex ways in which people benefit from forests, making efforts at achieving outcomes that benefit people as well as meeting biodiversity and carbon objectives ('win-win-win') much harder to achieve than is often admitted. Furthermore they highlight the entrenched political and economic asymmetries that pervade forest

conservation interventions, and have proved very difficult for such interventions to overcome. Since REDD+ is not being framed in a way that challenges these asymmetries, the risk of skewed distributions of costs and benefits in REDD+ remains high (also see Chapter 5). Table 4.3 provides an overview of the main insights from the chapter, highlighting economic and social risks and opportunities associated with previous interventions, and suggesting ways in which some of the risks can be mitigated.

Lessons from previous management actions for REDD+					Table 4.3
Management actions	Impacts across different stakeholders				Options to mitigate risks
	Economic impacts		Social impacts		
	Risks	Opportunities	Risks	Opportunities	
Addressing agricultural drivers	Loss of ecosystem service (ES) due to rebound effect; corruption during land transfers; land grabs.	Conservation of ES (when land sparing occurs); increasing domestic food budget; improving local economic growth; poverty reduction, emergence of new businesses; job creation.	Farmers may be forced to work against their will; farmers may be misinformed about their choices and corresponding benefits; some jobs may be lost; traditional agricultural livelihoods might be lost.	Conservation of ES (when land sparing occurs); development of new social infrastructure; job creation.	Clarification and security of land tenure, access and use; corruption control; extension services; labour-intensive practices; retraining programmes.
Protected areas	Possible conflict with development; reduced access; burden of costs; increased inequality (if elites capture benefits of jobs/tourism); loss of alternative land-use opportunities.	Conservation of ES; additional income from jobs, tourism or direct payments for conservation; avoided costs of deforestation.	Reduced access; centralised governance; displacement of homes or livelihoods; loss of cultural recognition, right to self-determination and identity (if displaced).	Conservation of ES; links to local networks and external actors through PA management (especially with participatory governance); recognition of traditional knowledge.	Participatory and decentralised approaches; community-led governance; respect for indigenous rights; Free Prior and Informed Consent (FPIC); market approaches and direct payments.
Sustainable management of forests	Lack of local benefits – incomes, employment and profit share; long gestation (and uncertainty) associated with returns on investment; loss of alternative land-use opportunities; inequitable distribution of benefits.	Increased income at national level (royalties); increased local job creation and income opportunities; increased access to credit and other markets; improved local infrastructure (roads, communications); conservation of ES (when deforestation is avoided).	Loss of traditional control and authority; displacement of homes or livelihoods; lack of participation in decision making; loss of local capacity, knowledge and ecological practices.	Tenure security; connection to local networks and external actors; conservation of ES (when deforestation is avoided).	Enforcement of national forestry codes; certification schemes with economic and social safeguards; voluntary standards; environmental and social impact assessments; company-community partnerships; small-holder plantations.
Restoration, afforestation and reforestation	Loss of livelihood; loss of land; inequitable distribution of benefits; loss of jobs to external/expert workers; international land 'grabs'.	Jobs; improved infrastructure; local development and investment; new financial opportunities (direct payments, markets for forest products); restoration of ES.	Displacement; social conflict; Suppression of traditional way of life; health risks (e.g. from plantation practices; human-wildlife conflict); loss of arable land; changes in social balance (migrant workers).	Empowerment; development of new skills and expertise; valuation and recognition of indigenous knowledge; improved tenure arrangements; restoration of ES.	Organisation in groups/associations; capacity building; extension services; clarification and security of land tenure, access and use; alternative livelihood schemes; agroforestry schemes.

The evidence considered here emphasises how broad political, economic and institutional conditions shape the social and economic impacts of interventions across the landscape. Figure 4.2 illustrates the conceptual thinking that has informed the discussion in this chapter. It demonstrates the need to consider interventions across the landscape, both within the forest and outside (especially in the agricultural context). It highlights the critical importance of the key mediating factors – governance, institutions, tenure and property rights – in shaping the ways in which ecosystems provide benefits to human society. REDD+ interventions can be managed in a variety of ways, which are associated with social and economic risks and opportunities. If implemented in a socially- and economically-inclusive manner, positive feedbacks will enhance the quality of the intervention. However, if there are negative social and economic impacts, these might undermine the effectiveness of the intervention, and compromise carbon and biodiversity goals alongside these adverse impacts on people and their livelihoods.

Security of tenure and associated authority for local decision making are important factors that have been found to facilitate stronger environmental management as well as the realisation of livelihood benefits. Conversely, poor recognition of such rights has led to poor rural people's exclusion from important livelihood assets and erosion of self-determination. The use of markets, and market-like instruments increases risks for those who have insecure tenure. In the case of PES, this has been especially important, as formally recognised management rights or title have often been a precondition for participation in PES contracts, and therefore essential for one's ability to receive direct benefits from such schemes. Clear tenure arrangements are also critical for forest certification and restoration (Forsyth, 2009; Oviedo, 2005; Charnley, 2005; Chamshama and Nwonwu, 2004). Without clear rules and trust in a system of land, property and use rights, local populations may not have an incentive to invest in restoration. Communities may fear that planting by the government may implicitly signify land appropriation; they may fear that planting indigenous species may signify that they will no longer be able to harvest the restored land; they may also fear that without adequate mechanisms to transfer benefits they may not receive due payment for providing an ecosystem service (Pejchar et al., 2007; Chokkalingham et al., 2005; Kiss, 2004; Barr and Sayer, 2012).

Tenure security is also a key consideration when addressing drivers of deforestation in the agricultural context. When tenure is secure and there are appropriate enabling conditions, sustainable agricultural intensification can benefit small farmers, while also creating the possibility of land sparing for forest conservation. Indeed, it has been suggested that increasing smallholder productivity might be one of the most effective tools in reducing poverty (de Janvry and Sadoulet, 2010; Loayza and Raddatz, 2010). When tenure is insecure or institutions are weak, however, the so-called 'land grabbing' phenomenon and other irregularities related to land transfers have been shown to limit the benefits from increased agricultural

productivity, often resulting in the expropriation of lands from smallholders.

The details of institutional arrangements and design at the intervention-specific level are also significant. In the case of PES, for instance, the design of PES contracts (e.g. their duration, benefit distribution mechanisms, conflict and grievance mechanisms) can directly and indirectly shape the extent and nature of their impacts on social relationships and economic assets (Tacconi et al., 2010). Similarly, in the devolved forest management context and in forest certification, interventions that have invested in building capacity and transparency in the governance of resources and project finances, have demonstrated stronger equity outcomes (McDermott and Schreckenberg, 2009). The same can be expected in the context of REDD+.

The distribution of costs and benefits is particularly sensitive: it applies across stakeholders at the local scale (with different communities potentially being impacted in different ways; on-site and off-site effects) and across scales, from local to global (Chan et al., 2007; Buckley and Crone, 2004). The problem of elite capture of intervention benefits is a key concern for future REDD+ initiatives. The sections above have shown that communities are not inherently homogenous entities, and that the principles on which both benefit distribution and representation in decision-making processes are based, require careful attention. In a gendered context, evidence shows that women's interests are rarely monolithic, and it is important to explore multiple categories of group differentiation (gender, ethnic, religious, caste, age and wealth) in order to understand outcomes (Mwangi and Mai, 2011). While difficult to achieve, the inclusive distribution of authority is important, including downward accountability, as well as upward and horizontal accountability (Ribot et al., 2006; Sikor and Lund, 2009). Participatory approaches will be key to negotiated outcomes that are beneficial to different stakeholders as well as to securing carbon and biodiversity gains. The role of neutral third parties (NGOs, professional facilitators etc.) is particularly important in supporting these negotiations (Brown, 2005; McShane and Wells, 2004; Edmunds and Wollenberg, 2005).

Balancing short and long term costs and benefits is also a significant challenge: human well-being concerns tend to be over short timeframes while biodiversity benefits are generally reaped in the longer term (Chan et al., 2007). This is particularly true for forest restoration for example which often requires high initial investments but returns are slow to come through, making it difficult for rural communities to engage in restoration without external support (Pejchar et al., 2007). In some cases, financial compensation has been granted, particularly following the establishment of protected areas or the establishment of plantations, to attempt to remedy the loss of assets or livelihoods. As compensation follows displacement of a previous activity, it should not really be seen as a positive outcome, especially since evidence demonstrates that social costs remain high, and the experience with development-induced displacement more generally suggests that

Economic and social impacts of REDD+ management actions on different stakeholders within a landscape

Figure 4.2



communities rarely benefit in the long term (Cernea and Schmid-Soltau, 2006). A long timeframe also equates to the possibility of many changes in socio-economic and political variables, such as changes in government policies and changes in funding, which creates uncertainty (Brown, 2005; McShane and Wells, 2004).

Ultimately, outcomes will largely depend upon how well new initiatives under REDD+ are able to learn from past institutional and governance lessons, such as the ones flagged in this chapter. The challenge should not be underestimated; it is far from straightforward to genuinely alter the political and economic asymmetries that have so far sustained inequities and exclusion from important livelihood assets in REDD+ target countries.

4.7 Conclusions

1. The way in which REDD+ is implemented has significant impacts for social and economic outcomes. On the one hand, REDD+ carries the potential to generate substantial positive impacts. Firstly, the primary objective of REDD+, avoiding deforestation and forest degradation, can greatly benefit poor people, as this group often feels the impacts of forest loss disproportionately. Secondly, REDD+ might generate substantial financial flows. If a significant fraction of these funds directly or indirectly reach the rural poor, they might generate considerable benefits. On the other hand, the poor are also most vulnerable to changes in resource management and access that can be part of REDD+, and can have their already marginal livelihoods severely impacted by such interventions.
2. For REDD+ to benefit the poor effectively, it is important to prioritise social and economic objectives alongside carbon and biodiversity goals. Giving parity to social objectives is necessary if REDD+ is to address the social and economic mechanisms that produce inequitable outcomes.
3. Whether it is a pro-active pursuit of social objectives, or an attempt to avoid social harm, sensitivity to social/economic objectives is difficult, and requires real commitment, which includes a consideration of governance and institutional arrangements, local engagement and participation, finance and markets, and timeframe.
4. To minimise risks (avoid the most negative social and economic consequences), safeguards are important, and must be sensitive to, and include monitoring systems for, tracking social impacts, especially access, authority and distributional issues.
5. Evidence suggests that pursuing social objectives alongside REDD+ will not only make the process more equitable but will also increase the likelihood of achieving carbon and biodiversity goals.
6. Genuine 'win-win-win' outcomes are not always available and there are sometimes difficult trade-offs to be negotiated between carbon, biodiversity and social objectives. In these situations, a careful and inclusive evaluation should explicitly consider the following possibilities: (a) acknowledge the negative social/economic consequences, but do nothing about them; (b) compensate the losers (financially), but acknowledge and accept social losses and disruption; (c) compensate the losers, and invest in secure, alternative livelihoods to attempt to offset some social losses; (d) abandon the carbon/biodiversity projects. The choice amongst these alternatives is likely to reflect the values and beliefs of the decision maker(s). It will also reflect the extent to which social objectives have been mandated within REDD+ policy.
7. Security of tenure and associated authority for local decision support better environmental management, as well as the realisation of livelihood benefits. Tenure security includes recognition of all forms of ownership and control, especially communal tenure. Poor recognition of such rights excludes the rural poor from decision making, and denies them access to potential benefits from market-based interventions, such as PES and REDD+. Weak tenure security also facilitates 'land grabbing' and other irregularities related to land ownership and transfer, which typically result in expropriation of lands from the most vulnerable groups.
8. There is growing evidence that improving participation in decision making has positive impacts for equity and for environmental effectiveness. Participatory decision making typically involves some form of collaborative property regime, coupled with decentralised authority structures, which are supported by appropriate rules and institutions.
9. An integrated landscape management approach is a powerful tool to address and reconcile the many environmental, social and economic aspects relevant to REDD+ inside and outside forests. REDD+ interventions span agricultural areas and forests, and thus interact with a multitude of needs and aspirations held by very diverse stakeholders. Careful and inclusive spatial planning can positively influence the distribution of winners and losers across the landscape so that REDD+ acts in the interests of the most vulnerable groups.
10. Significant knowledge gaps remain in a number of areas that are important for understanding the social and economic consequences of REDD+ and biodiversity strategies. These include:
 - Information about the impacts of forest loss and degradation on the livelihoods of the poor. There is still limited large-scale comparative research on the social costs of forest loss and degradation, and how these are distributed amongst diverse rural populations. As a corollary, there are few studies that systematically document costs and benefits to local communities stemming from forest restoration. While small-scale

anecdotal examples exist and some research focuses on the economics of restoration, there is as yet little experience of documenting the range of social benefits and costs to stakeholders in a diversity of settings.

- Knowledge about the dynamics of agricultural intensification remains limited. For instance, there is little evidence about the precise set of conditions and approaches, in a variety of contexts, that would make land sparing dominant over the rebound effect following agricultural intensification. We also lack information about the conditions under which agricultural intensification is more likely to lead to job losses, and the specific policy approaches that could mitigate or reverse this impact.
- Knowledge of when and where decentralisation and participation are appropriate for ecosystem management, and the conditions that favour success in protected areas, sustainable use strategies and restoration, has progressed well. Less is known about how a global REDD+ architecture can fit into documented best practice, for instance because it tends to expect upwards accountability towards funders and the market, rather than downwards accountability to local stakeholders. Systematic studies that draw lessons from project level REDD+ pilots need to build evidence about these important governance and institutional issues.
- There are some important lessons emerging from studies on PES, especially from more qualitative studies of the process of implementation. Much of the evidence is limited to incentivising desired management practices on private farmlands, and there is insufficient evidence of how behaviour might change when similar strategies are implemented at a large scale on state or communal land. Thus, there is limited knowledge about how incentives within a PES-based REDD might function in conjunction with, for instance, protected area management.
- There is little systematic monitoring of the flows of benefits and costs to diverse stakeholders from ecosystem interventions (agricultural intensification, protected areas, sustainable use strategies, restoration, reforestation and afforestation) across a variety of landscapes. There is considerable scope for monitoring to be incorporated into strategies that tackle multi-dimensional poverty and to provide more evidence about how existing or restored ecosystem services interface with attempts to improve the quality and resilience of rural livelihoods.
- There is also relatively little information about what local people in different places consider to be fair and beneficial. While it is relatively straightforward to offer material compensation for economic losses, there is limited knowledge about how people perceive socio-cultural impacts on lives and livelihoods, and this poses risks for REDD+ contexts that involve potential trade-offs with the interests of local stakeholders (in addition to normative concerns). Knowledge in this area is likely to be helped by an emerging body of research into well-being, which is progressing the understanding of needs, rights and issues of justice, equity and fairness in ecosystem management.
- Further research is needed on decision-making methods and tools that help to consider and incorporate the interests of diverse stakeholders involved in or impacted by ecosystem-based interventions. There is a need to understand and draw general lessons from practical integrated landscape management approaches that seek to create synergies between multiple ecosystem services and the interests of diverse stakeholders.

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Chapter 5

Governance for REDD+, forest management and biodiversity: Existing approaches and future options

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Abstract: The chapter examines the evolution of REDD+ governance and identifies policy options to increase synergies among REDD+, the sustainable management of forests and biodiversity conservation. REDD+ emerged at the international level as a point of convergence across the ‘institutional complexes’ of forests, climate and biodiversity. This convergence attracted the engagement of a wide range of institutions in REDD+ activities, which together have drawn on three primary sources of authority to influence REDD+ rule-making: government sovereignty, contingent finance and voluntary carbon markets.

Intergovernmental processes, which represent the primary articulation of governmental authority at the global level, have generated few binding commitments to the sustainable management of forests or biodiversity due to conflicting country interests. These efforts instead have favoured normative guidance, monitoring and reporting, and legality verification initiatives that reinforce sovereign authority. Bilateral and multi-lateral finance initiatives have exerted ‘fund-based’ authority through the application of operational safeguards protecting indigenous and local communities and biodiversity, but limited funding and low capacity of REDD+ countries to absorb those funds have constrained their influence. Finally, non-state actors have developed voluntary certification schemes for forest and carbon as a ‘fast track’ approach to elaborating more substantive international standards for environmentally- and socially-responsible forest practices. While the small size and voluntary nature of markets for forest carbon have greatly constrained the impact of these approaches, this could change if a significant regulatory market for REDD+ develops.

Furthermore, the governance of REDD+, forest management and biodiversity is pluralistic, involving multiple institutions and actors. Efforts to promote REDD+ safeguarding at the international level exist in tension with national sovereignty and local autonomy. This complexity is taken into consideration in the suite of policy options provided in this chapter, which suggest the need to draw on a range of institutions and approaches and to consider how together they influence the balance of power and incentives across actors and scales.

5.1 Introduction

REDD+ interventions occur within a broader multi-level *governance*¹ landscape that shapes *forest* and *biodiversity* outcomes. This chapter examines existing and potential governance and policy approaches for REDD+, and how they complement or contradict efforts to sustainably manage forests and conserve biodiversity in a manner that enhances social cohesion and welfare. This includes an analysis of the emergence of REDD+ within a broader landscape of international, national and local governance, and the insights this provides regarding which actors and institutions are best positioned to integrate multiple objectives into REDD+.

As observed in Chapter 4, understanding the socio-political context of REDD+ rule-making is critical for understanding how various REDD+ interventions are likely to play out in ‘real world’ settings. While Chapter 4 focused on the socio-economic dynamics of forest and land use and their implications for REDD+, this chapter looks at REDD+ interventions and their interplay within broader governance contexts.

Our analysis builds upon a growing body of literature on the governance of REDD+. Due to the newness of REDD+ and lack of empirical data on its effectiveness, much of this literature has focused on learning lessons from forest governance (e.g. Angelsen et al., 2009; Kanowski et al., 2011), on identifying normative principles for good or effective governance (e.g. Sikor et al. 2010; Lyster, 2011) and, increasingly, on examining case studies of early REDD+ interventions (Angelsen et al.,

2012). Common to these studies is the awareness that the design of REDD+ is an inherently political act (Skutsch and McCall, 2010; Thompson et al., 2011), involving different actors with different interests and ideas vying for the authority to write the rules (Angelsen et al., 2012). However, while much of this past literature is cognizant of the power dynamics inherent in REDD+ decision-making, there is a lack of analyses that ground discussion of policy options for REDD+ in the consideration of which actors and institutions hold the power to achieve particular desired outcomes. Such grounding is critical for examining policy options that serve environmental and social objectives which lie outside the core framing of REDD+ as a mechanism for reducing forest *emissions*. As discussed in Chapter 4, where these objectives are treated as peripheral to emissions reduction, there is much uncertainty about how they might be addressed. The emphasis of this chapter is thus on what can, and cannot, be done at different scales, by which actors, to create an integrative REDD+.

Section 5.2 focuses at the international level, examining the emergence of REDD+ in the climate regime and evolving options for international governance of REDD+, forest management and biodiversity. Section 5.3 considers the intersection of international governance with national and local agendas, and conflicting pressures for international standardisation, sovereignty and local autonomy, illustrated by case study boxes from the Congo Basin, Indonesia, Nepal and Brazil.

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

5.2 International governance

There is a wide and growing array of international institutions of relevance to the governance of forests, *carbon* and biodiversity. This phenomenon of institutional diversity in international environmental governance has sparked an ever-increasing body of literature about the ways in which institutions interact, the consequences of interactions, and ways of managing those consequences (e.g. Young, 1996; Rosendal, 2001; Stokke, 2001; Oberthür and Gehring, 2006; Oberthür and Stokke, 2011). Informed by this literature, we adopt the term ‘institutional complex’ to refer to the cluster of institutions associated with a specific issue area. International institutions of potential relevance to REDD+ range from those focused on the environment to those related to trade and human rights. Our emphasis is on institutions focused on the substantive areas of ‘forests’, ‘climate’ and ‘biodiversity’, and their overlap with REDD+, which essentially forms a ‘sub-complex’, as portrayed in Figure 5.1 below.

5.2.1 A brief history of international forest, climate and biodiversity governance, pre-REDD+

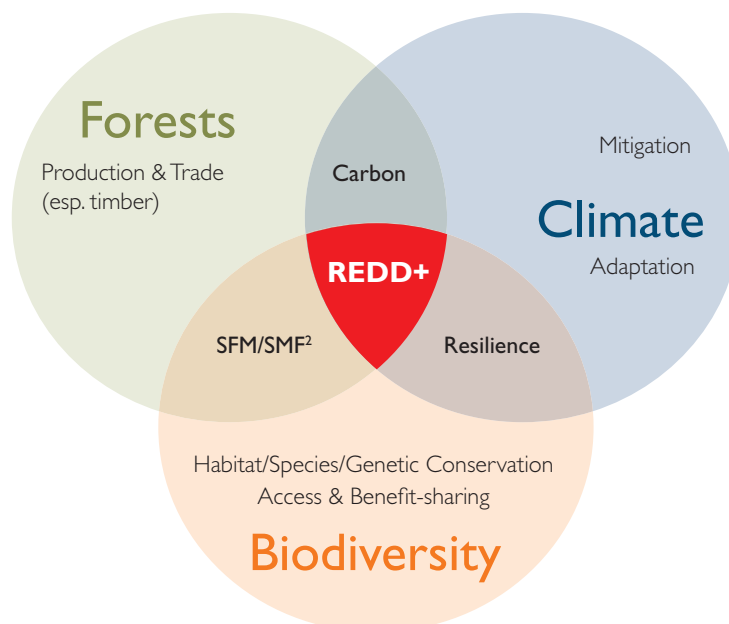
Early beginnings of the international institutional complex on forests can be traced back to 1946, with the launch of a global forest inventory by the Food and Agricultural Organization of the United Nations (FAO). The

FAO’s monitoring efforts contributed to growing global awareness of tropical forest loss, which in turn spurred two intergovernmental tropical forest initiatives in the 1980s – the International Tropical Timber Organization (ITTO) and the Tropical Forestry Action Plan (TFAP). While agricultural expansion, not forest production, was the leading cause of tropical *deforestation* (Geist and Lambin, 2002), both ITTO and TFAP had little mandate to reach beyond the forest sector. Instead, their focus was on ‘*sustainable forest management*’ (SFM), a broad concept encompassing timber production, biodiversity conservation, livelihood concerns and other complementary objectives. The exclusive focus of both ITTO and TFAP on tropical forests also meant that participating tropical countries were reluctant to make commitments to forest conservation in the absence of similar commitments from temperate and boreal countries (Humphreys, 2006). This led to proposals to launch a global forest convention at the 1992 Earth Summit in Rio de Janeiro.

At the Earth Summit, divergent country interests prevented consensus on a global forest convention. However, two other conventions were adopted that are of central relevance to the climate and biodiversity complexes – the United Nations Framework Convention on Climate Change (UNFCCC) and the Convention on Biological Diversity (CBD). In comparison to the scope of forest negotiations, the UNFCCC’s focus on stabilising atmospheric *greenhouse gas* concentrations is quite narrowly defined – arguably facilitating intergovernmental

Overlapping institutional complexes of core relevance for REDD+, forest management² and biodiversity³

Figure 5.1



² The phrase ‘sustainable forest management’ (SFM) is commonly used within the forest complex while the term ‘sustainable management of forests’ (SMF) is used within the biodiversity and climate complexes.

³ The sub-themes in grey text list the issue areas that have emerged as major foci in each institutional complex. The list is neither exhaustive nor are most issues exclusive to a particular complex.

consensus. The Kyoto Protocol, adopted in 1997, operationalised this objective by specifying binding emission reduction commitments for industrialised countries for the period 2008-2012. The 2011 climate conference in Durban arguably softened the ‘firewall’ between developed and developing country commitments in view of a future climate agreement, referring to “mitigation efforts by all parties” (UNFCCC, 2012a: para. 7), but it remains to be seen how a new burden-sharing under an agreement that is applicable to all parties will translate into practice (e.g. Rajamani, 2012).

The scope of the CBD straddles that of the forest and climate change processes. Its core objectives are: 1) the conservation of biodiversity, 2) the sustainable use of its components, and 3) the fair and equitable sharing of benefits from the utilisation of genetic resources (Article 1 CBD). Agreement on the third objective on benefit-sharing was crucial to securing the support of tropical forest countries, giving them the opportunity to gain revenue from the commercial exploitation of their biodiversity. The benefit-sharing objective was also supported by non-governmental organisations (NGOs) who wished to see some of the commercial benefits from biodiversity exploitation flowing to the community level (McNeely et al., 1995). Non-governmental organisations have been active participants in the CBD, and a driving force behind its core strategies (Arts, 1998), which include soft targets for expanding protected areas, arresting species loss and access to, and benefit sharing of, the utilisation of local and *traditional knowledge*. However, governments have generally not backed these aspirational goals with legally binding commitments. Parties are asked to establish their own priorities, in this case via National Biodiversity Strategies and Action Plans (NBSAPs) that translate global goals into nationally/appropriate actions.

While the CBD thus remained limited in its authority to command government action, its efforts were bolstered by various scientific initiatives which were launched with major NGO involvement. These include the International Union for the Conservation of Nature’s (IUCN) Red List of Threatened Species, established in 1963, and the World Database on Protected Areas, first launched as an independent non-profit venture in 1988. The initiatives support the assessment of progress towards the global targets and country reporting called for under the CBD.

As the institutional complexes for climate and biodiversity thickened, intergovernmental forest negotiations entered into a period of relative stalemate. Factors impeding agreement on a forest convention included differences in the negotiating power of a country’s domestic timber industry, differences in country dependence on international trade, and disagreement as to whether developed countries should transfer finance and technology to tropical forest countries in exchange for conservation commitments from the latter (Humphreys, 2006). As a result, countries varied in their willingness to relinquish sovereign authority on forest management to an expanding array of international norms for sustainability (Dimitrov, 2005). Non-governmental actors were initially supportive of a convention but later withdrew support for

fear that countries with powerful timber industries would dominate the process resulting in low standards for forest protection (Humphreys, 1996).

Hence, intergovernmental forest negotiations have generated exclusively ‘soft law’, that is agreements on non-legally binding principles and processes, such as the 1992 ‘Forest Principles’, Chapter 11 of Agenda 21 on deforestation, and the 200+ proposals for action produced by the Intergovernmental Panel on Forests (IPF) and its successor, the Intergovernmental Forum on Forests (IFF) (Humphreys, 2006). These processes have also institutionalised a system of National Forest Programmes that, like the CBD’s NBSAPs, are intended to encourage countries to establish their own national goals and priorities. In addition, the United Nations Forum on Forests (UNFF) – which succeeded the IPF and IFF – adopted a Non-Legally Binding Instrument on All Types of Forests in 2007, reflecting general principles and points of convergence among all participating countries. Although these efforts may provide normative pull, as well as facilitate coordinated global monitoring and reporting, the degree to which they do so depends profoundly on their (voluntary) uptake within individual countries.

The slow pace and limited commitments sparked the launch of an alternative approach that turned to markets as a potential source of international authority (Cashore et al., 2004). In 1993, several NGOs and a collection of timber buyers and retailers launched the Forest Stewardship Council (FSC) as a non-state, market-driven instrument designed to incentivise sustainable forest production through the green labelling of timber products. The FSC was created after the ITTO had declined to implement a labelling scheme, indicating the ongoing conflict among interests in the forest sector. As further evidence of conflicting interests, competing national certification schemes emerged in the following years, supported by forest producers’ associations in Europe, North America and elsewhere (Auld et al., 2008). Each of these schemes has developed its own set of standards for SFM, highlighting the contested nature of the concept. Many schemes are now consolidated under the Programme for the Endorsement of Forest Certification (PEFC - also see Chapter 4).

By the 2000s, the issue of ‘illegal logging’ began to re-energise intergovernmental negotiations, this time at a regional level. The sub-global scale of these efforts, and their focus on legality rather than sustainability, has been heralded as a major breakthrough, due to the smaller number of negotiating parties, the restricted scope but expanded scale and the promise to strengthen rather than undermine national sovereignty (Bernstein et al., 2011). Tackling illegal harvesting is attractive to several actors: environmentalists see it as a means to reduce the environmental damage of logging practices and to promote more responsible global consumption; host governments see it as a means to strengthen sovereignty and increase tax revenues; the legal timber industry sees it as a means to increase their competitiveness.

Subsequently several regional processes emerged strengthening forest governance in Africa, Asia and Eurasia. The EU Forest Law Enforcement, Governance and

Trade (FLEGT) process integrated supply-side efforts to stem illegal logging with demand-side measures aimed at restricting the imports of illegal timber products into the EU. This was to be achieved through bilateral ‘Voluntary Partnership Agreements’ (VPAs) between the EU and participating developing countries. Currently six VPAs have been signed⁴. The effectiveness of these VPAs in reforming forest governance as well as their impact on the sustainability of forest practices remains to be seen.

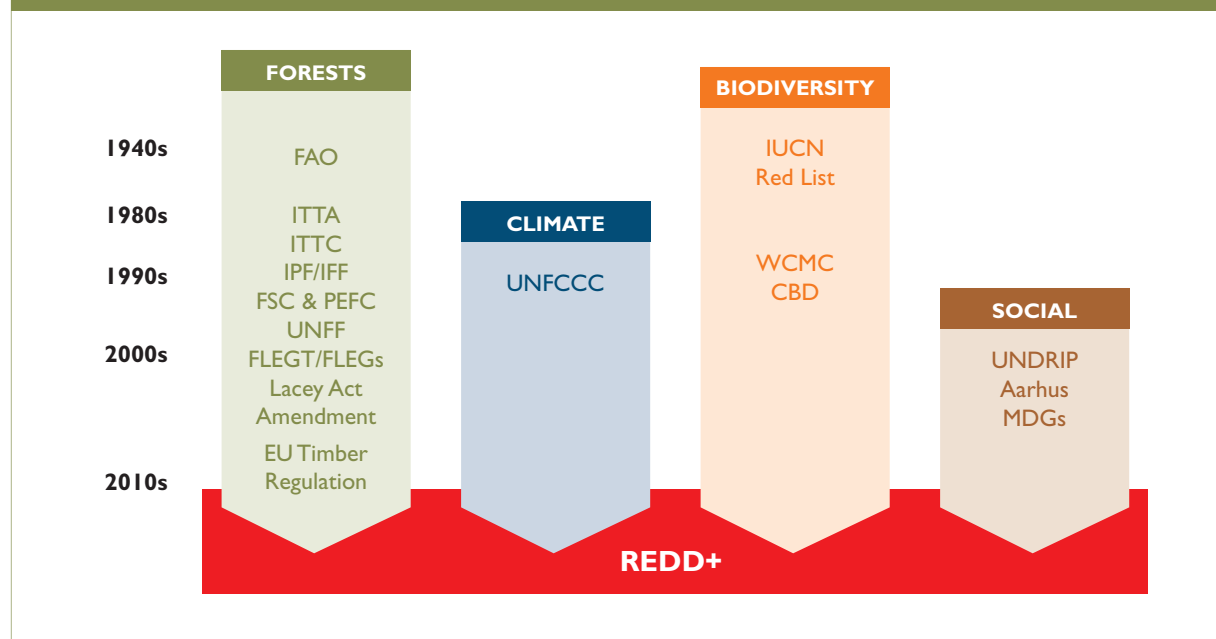
Around the same time, many governmental and non-governmental actors were sharpening their focus on the ‘social’ dimensions of international governance. The 1998 Aarhus Convention established norms for public participation in environmental issues. In 2000, the UN Summit addressed international concern for rising global income disparities by adopting eight Millennium Development Goals (MDGs) including non-binding targets to eradicate extreme poverty. In 2007, the UN General Assembly adopted the UN Declaration on the Rights of Indigenous Peoples (UNDRIP). UNDRIP enshrined the principle of ‘free, prior and informed consent’ (FPIC), which asserts the right of indigenous peoples to block activities that impact their traditional lands and practices. While providing strong normative signals, UNDRIP is a non-legally binding declaration subject to national interpretations, and lacking intergovernmental mechanisms for enforcement. Nevertheless, these initiatives together strongly legitimate and institutionalise the integration of social concerns into environmental rule-making.

Figure 5.2 provides a summary timeline of the key institutional developments in the forest, climate, biodiversity and social arenas.

The sum of these evolving instruments, agreements and processes emerging at different scales and involving different actors amounts to a fairly comprehensive, overlapping and sometimes conflicting international governance complex (McDermott et al., 2011). Notably, early efforts at international coordination were largely concentrated in intergovernmental processes. Within this sphere of ‘government-based’ authority, disagreement on the appropriate balance of priorities for forest management and biodiversity in particular, precluded agreement on binding commitments and favoured instead actions with potential to enhance sovereignty, including global monitoring, national-level planning, target-setting (usually voluntary) for the more narrowly defined objectives (e.g. emissions, protected areas) and legal enforcement. In general, enforceable commitments have been achieved more readily for the singular goal of emissions reductions than for the broader and less readily measured goals of sustainability and biodiversity conservation. Non-governmental organisations, seeking to push international standards for forest management and biodiversity protection beyond government willingness to do so, have pursued certification as a voluntary market-based approach. Certification aims to draw power and authority from market demand for environmentally- and socially-‘responsible’ forest products. By-passing government resistance, NGO-driven certification schemes were able to create relatively stringent requirements, but have been subject to competition from conflicting industry-driven schemes. Meanwhile, the reach of certification’s influence is limited by a lack of market demand in developing countries.

A summary timeline of the key institutional developments in the forest, climate, biodiversity and social arenas

Figure 5.2



⁴ http://www.euflegt.efi.int/portal/home/vpa_countries/ [Accessed on: 2 July 2012]. The six countries are Cameroon, Central African Republic, Ghana, Indonesia, Liberia and Republic of Ghana.

5.2.2 The emergence and proliferation of REDD+ governance

REDD+ emerged within the UNFCCC as a mechanism to create financial incentives for contributing to *mitigation* of forest-related greenhouse gas emissions by developing countries. This emphasis on financial incentives was central in overcoming initial barriers to the inclusion of forests in the climate institutional complex. It also spurred a further proliferation of actors, institutions, and sources of authority engaged in international forest and biodiversity governance, as will be seen from the following historical account.

Allowing industrialised countries with emissions reduction commitments to use land-based greenhouse gas removals to offset their emissions was hotly contested during the negotiations on (and following) the Kyoto Protocol. The concern was that accounting for forest carbon would relieve pressure on these countries to reduce fossil fuel emissions. In addition, there were a range of technical concerns, such as problems of permanence (the risk of forest loss and reversal of climate benefits), *leakage* (the risk of displacing forest loss in one region to another region) and carbon accounting (difficulties in accurately measuring changes in forest carbon stocks) (Noble and Scholes, 2001). It was agreed that industrialised countries could use land-based removals as offsets up to a certain limit. In addition, under the Clean Development Mechanism (CDM), industrialised countries were allowed to use offsets through *afforestation and reforestation* projects they supported in developing countries. However, reduced (or avoided) deforestation was not included in the CDM. Many NGOs and some indigenous groups were dismayed at this outcome, fearing that the design of the CDM, which focuses solely on the carbon sequestration role of forests, would run counter to the conservation of biological diversity (Streck and Scholz, 2006).

At the eleventh Conference of the Parties to the UNFCCC in 2005, Papua New Guinea and Costa Rica retabled the discussion by presenting options for reducing emissions from deforestation in tropical countries under a post-2012 climate regime. Negotiations led to several decisions on REDD+, the most important of which (so far) has been the Cancun Agreements (UNFCCC, 2011). The Cancun Agreements establish that participation in REDD+ is voluntary and national government-driven, unlike the project-level CDM. In this way, REDD+ would first and foremost be governed, implemented and measured at the national level. Such a national approach appeared to mitigate concerns around leakage and accounting that plagued forest negotiations around the CDM, as it would capture the domestic (if not the international) displacement of emissions.

Further contributing to developing country support, the '+' in REDD+ has been added to reflect the inclusion of forest conservation, forest management and forest carbon stock enhancement. This move was critical, first, to denote that REDD+ is concerned with the broad range of forests goods and services and not just carbon and, second, to gain the support of countries with constant

or increasing forest cover such as China and India (Potvin and Bovarnick, 2008). REDD+ was to occur in three phases to accommodate differing country capacities, starting with national planning and 'readiness' (phase 1), followed by the implementation of national strategies (phase 2) and, eventually, full accounting against national reference scenarios (phase 3) (see Figure 5.3). Finally, activities would depend on developed countries providing adequate financial and technical support throughout all phases, which in phase 3 means providing financial incentives to developing countries for the reduced emissions measured in changes of forest carbon against national baseline or reference (emission) levels.

In addition to addressing a range of concerns relating to sovereignty, accounting and finance, the Cancun Agreements addressed social and environmental issues of central importance to many NGOs and other actors. Specifically, Appendix I of the text contains language on social and environmental 'safeguards' that must be respected whilst implementing REDD+ activities (see Box 5.1, Section 5.2.3). The Appendix echoes objectives from the various multilateral processes discussed above, ranging from addressing the drivers of deforestation, to governance, poverty alleviation, participation, indigenous rights, the conversion of forests to *plantations*, and biodiversity conservation. However, and as discussed further in Section 5.2.3.1, it remains unclear what constitutes adequate safeguarding or how countries will be held accountable for achieving it.

Despite the relative progress in REDD+ negotiations, there are many issues that remain undecided and vague, such as: rules for establishing REDD+ baselines of performance (reference levels); monitoring, reporting and verification (MRV); and international accountability for safeguarding. Moreover, regardless of progress made in the negotiations, the lack of stable, predictable sources of finance for REDD+ threatens its longer-term viability (Streck and Parker, 2012).

While international negotiations drag on, decision-making on REDD+ has proliferated beyond the UNFCCC to multiple arenas, from the preparation of 'guidance' notes for REDD+ under the CBD, to the emergence of a voluntary carbon market for REDD+ projects, to various multilateral and bilateral initiatives outside of the global climate regime. The patterns of this proliferation offer clues as to what types of international institutions beyond the UNFCCC – intergovernmental, private, regional, bilateral, etc. – may carry authority to address different dimensions of REDD+ (Korhonen-Kurki et al., 2012).

Given the basic logic of REDD+ as a financial incentive mechanism, financial institutions and the power and liability they hold, have emerged as a new and core source of rule-making for REDD+ actions, which we refer to as 'fund-based' authority. In particular, several global, multilateral financing initiatives have played a key role in supporting REDD+ 'readiness' activities in over 40 countries. One of these, the World Bank's Forest Carbon Partnership Facility (FCPF), was launched at the 2007 session of the UNFCCC Conference of the Parties in Bali to help countries prepare for REDD+, and to provide technical and scientific support with respect to issues such as MRV

and the achievement of ‘multiple benefits’. The FCPF is a partnership of developing and developed country governments that also includes private sector representatives and NGOs. It serves the dual goal of building capacity for implementing REDD+ in developing countries through the establishment of national monitoring systems, management systems and stakeholder consultation arrangements (through its Readiness Fund), and testing the feasibility of performance-based payments through pilot activities (through its Carbon Fund). Another World Bank initiative, the Forest Investment Programme (FIP), also seeks to build capacity, and aims to support national policies and measures to implement REDD+. In 2008, the UN-REDD programme was created by three UN agencies – FAO, UNEP and UNDP – to complement the efforts of the FCPF and bilateral initiatives (UN-REDD, 2008). UN-REDD supports REDD+ readiness activities, strengthening governance and stakeholder participation and supporting local capacity-building. A large focus is on MRV, using FAO expertise and its networks in 194 member countries. In addition to these new funding initiatives, existing financial mechanisms have also included REDD+. For instance, the Global Environment Facility (GEF) has started to address REDD+ in its fifth replenishment, in part in response to developments in the UNFCCC (GEF, 2010). Although the roles of the GEF (phases 1-3) and FIP (phase 2) are clear in theory, their relevance for stakeholders on the ground remains to be clarified (Hardcastle et al., 2011). The Congo Basin Forest Fund and the Amazon Fund are two examples of regional funding mechanisms for forest protection and sustainable management that are administered by regional banks according to their own rules and procedures.

In addition to multilateral initiatives, individual donor governments have become active. Norway, in particular,

has concluded bilateral agreements with Brazil, Guyana and Indonesia, promising large REDD+ payments dependent on demonstrated reductions of deforestation from agreed reference levels. The country thus plays a pioneering role in testing phases 2 and 3 of REDD+ implementation.

These various initiatives are governed by laws and policies including, in the case of the multilateral financial institutions, distinct sets of environmental and social ‘safeguards’ to protect their investors from risk. By making funding contingent on meeting these safeguards, they hold the power to enforce them. Unlike for the UNFCCC, the challenge for these initiatives thus lies less in the definition of enforceable safeguards but rather in the limited capacity of recipient countries to meet the requirements and ‘absorb’ the funds available in a timely manner (Nussbaum et al., 2009) - a problem exacerbated by the overlap of requirements across financing institutions.

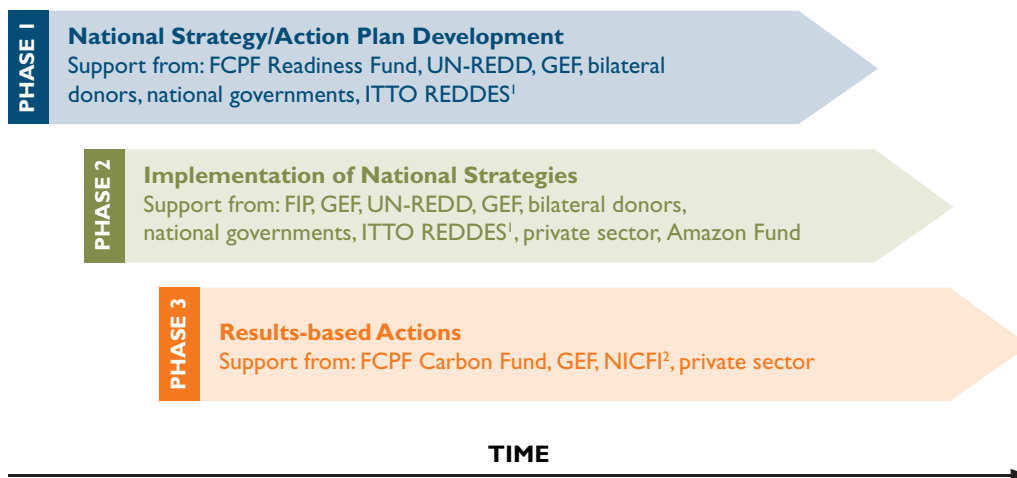
Figure 5.3 shows that the different initiatives seek to cover activities in each of the three REDD+ phases, although most of the funding so far has targeted phase 1 – and to a lesser extent phase 2 (Agrawal et al., 2011), while no country has yet reached phase 3.

Coordination among the multilateral funding initiatives has improved over time (Hardcastle et al., 2011) and takes place, for instance, through coordinated responses to proposals for funding, and the joint provision of supporting services to the (interim) REDD+ Partnership. The latter is an intergovernmental platform established at the Oslo Climate and Forest Conference in May 2010, which is seen as a forum for knowledge-sharing and learning on REDD+.

In parallel with these financing arrangements and their overlapping requirements, other actors, including sub-national governments, have been active promoting

Potential financial support for three phases of REDD development (adapted from Hardcastle et al., 2011)

Figure
5.3



¹ Thematic Programme on Reducing Deforestation and Forest Degradation and Enhancing Environment Services

² Norway's International Climate and Forest Initiative

market-based approaches for incentivising and governing REDD+. One notable trans-sub-national initiative is the Governors' Climate and Forests Task Force, which brings together 16 states and provinces from Brazil, Indonesia, Mexico, Nigeria, Peru and the United States. The Task Force seeks to link REDD+ activities in tropical forest countries to recently adopted climate change legislation in California, thereby paving the way for a regulated REDD+ carbon market (Agrawal et al., 2011).

Meanwhile, conservation and development NGOs, as well as the private sector, have started to implement a host of REDD+ pilot and demonstration activities on the ground. A recent review counts at least 100 such demonstration activities globally (Cerbu et al., 2011). Echoing strategies in the forest sector, NGOs have become increasingly involved in developing environmental and social standards and schemes for certifying REDD+ projects and the carbon credits associated with them (Merger et al., 2011). All of these efforts bear evidence to the emergence of market-based governance as a significant source of authority steering REDD+ activities.

In sum, REDD+ emerged in the intergovernmental arena – i.e. the UNFCCC – with its main focus on reducing emissions, coupled with requirements to monitor and report on very broadly defined 'safeguards' echoing other intergovernmental agreements. Fund-based and voluntary market-based institutions have stepped in with operationally-defined safeguards. These respond either to concerns about investor risk or to the desire to promote particular environmental and social values. The former are addressed through the institutions' authority to withhold funds, while incentives such as greater market share or price premiums for certification seek to stimulate desirable REDD+ activities.

5.2.3 Options to synergise climate, forest management and biodiversity objectives at the international level

This section delves in more detail into the governance mechanisms adopted by the evolving REDD+ initiatives discussed above, and examines existing and potential approaches for integrating forest management and biodiversity objectives. The analysis is organised around the three key spheres of authority that shape international forest, climate, biodiversity and REDD+ governance to date: governmental (based on sovereign authority), fund-based (rooted in direct control of financial flows) and market-based (rooted more diffusely in market demand). While each of these spheres is analytically separate, they interact in important ways within and across institutions. For example, governments may exert fund-based authority through the conditional provision of finance for REDD+. The analysis therefore identifies the primary source of authority driving different approaches, while acknowledging that no single source operates in isolation.

5.2.3.1 Governmental authority

Drawing on our historical analysis above, intergovernmental processes are likely to contribute to synergies

among climate, forest management and biodiversity objectives through three primary pathways: 1) the provision of (mostly voluntary) normative guidance, including a limited number of narrowly/defined targets; 2) catalysing and coordinating monitoring and reporting; 3) legal trade restrictions aimed at reinforcing within-country legal compliance (e.g. FLEGT).

Normative guidance

Under the UNFCCC, social considerations, biodiversity and forests are covered in Appendix I of the Cancun Agreements in the form of guidance and safeguards. The guidance acknowledges goals of other international forums by calling on countries to take into account the multiple functions of forests and other *ecosystems* (echoing the UNFF) and to implement REDD+ in the context of sustainable development and reducing poverty (echoing the MDGs). It also spells out seven safeguards that should be promoted and supported when undertaking REDD+ activities (Box 5.1).

Like the guidance text, these safeguards also reiterate and/or mirror goals and mechanisms from other intergovernmental processes, including national forest programmes under the UNFF, governance (forest law enforcement and governance (FLEG) processes, respect for the rights of indigenous peoples (UNDRIP), participation (Aarhus Convention) and the conservation of biodiversity (CBD). Decisions taken by all of these different institutions carry normative relevance, if not legal force. Among them, the CBD has been particularly pro-active in developing guidance for REDD+, even though this guidance has not been formally solicited by the UNFCCC (van Asselt, 2012; see Annex B for details on the content of this advice). Researchers and activists have also pointed to the significance of the application of UNDRIP's FPIC as a prerequisite for all REDD+ activities (Anderson, 2011). Regional-level processes can also provide synergies, whether through regional coordination of REDD+ activities or through complementary efforts such as FLEGT (see Annex C for an example from Central Africa). However, while there appear to be many opportunities for

REDD+ Safeguards (UNFCCC, 2011: Appendix I, para. 2)

Box
5.1

- Consistency with the objectives of national forest programmes and relevant international conventions and agreements;
- Transparent and effective national forest governance structures;
- Respect for the knowledge and rights of indigenous peoples and members of local communities;
- Full and effective participation of relevant stakeholders;
- Consistency with conservation of *natural forests* and biological diversity;
- Actions to address the risks of reversals;
- Actions to reduce displacement of emissions.

these intergovernmental processes to work together, the different actors and interests involved to date show limited cross-sectoral cooperation (Rayner et al., 2011).

Defining appropriate national monitoring and reporting systems

The Cancun Agreements request developing countries engaging in REDD+ to develop a national forest monitoring system and a system for providing information on how the various safeguards listed in the decision are being addressed and respected throughout the implementation of REDD+ activities, taking into account national circumstances and capacities, recognising national sovereignty and legislation and relevant international obligations and agreements, and respecting gender considerations (UNFCCC, 2011). The same decision also requests the UNFCCC Subsidiary Body for Scientific and Technological Advice to develop a work programme, including on guidance for establishing such information systems on applying safeguards.

In keeping with a country-driven approach, the UNFCCC has thus far not linked its safeguard text with international performance standards or mechanisms for verification. Given the broad scope and political implications of the safeguards, evidence from past intergovernmental processes suggests it is unlikely that countries will agree to binding commitments that limit their sovereignty on these issues (e.g. Lee et al., 2011). Instead, and consistent with past processes, the emphasis has been placed on country-designed monitoring and reporting (UNFCCC, 2012c). Monitoring and reporting of REDD+ safeguards is likely to be integrated into the new process for international consultation and analysis that will be required of the biennial update reports on emission trends from developing countries. This may spur some degree of standardisation and the possibility of independent monitoring. However, without internationally-enforceable performance thresholds it will be up to individual countries to define adequate performance. Therefore, even if countries reached consensus on independent monitoring, it would be restricted to verifying information rather than evaluation.

Legal trade restrictions

The increasing participation of countries in Africa, Asia and Latin America in FLEGT and other illegal logging initiatives suggests that intergovernmental agreement may be relatively easily attained for measures aimed at strengthening the ability of participating countries to enforce their own laws (in the case of developing countries) and/or protect their industries (in the case of developed countries). Reinforcing these trends is a growing number of timber procurement policies that require governments in importing countries to verify the legality or sustainability of the timber they purchase. Likewise, the recent expansion of the Lacey Act in the US and the passage of the EU Timber Regulation are new policies that prohibit imports of wood products produced in violation of the rules of their country of origin. Governments have also shown interest in applying similar legality measures to key agricultural crops driving deforestation – e.g. palm

oil and soy (e.g. UK, 2004). For those countries reliant on exports to the US or EU, these kinds of initiatives may help reinforce existing laws protecting forests, biodiversity and local communities. However, such approaches may do little to incentivise countries where domestic or other foreign markets are the primary drivers of deforestation, and/or which lack robust environmental and human rights laws. Furthermore, if proof of legality creates a significant barrier to trade, then countries may be incentivised to lower their environmental standards to ease verification requirements and improve their global competitiveness.

5.2.3.2 Fund-based governance

Linking implementation of safeguards to the allocation and distribution of REDD+ finance is arguably the most powerful lever for asserting international priorities for co-benefits under REDD+ for several reasons: 1) it provides direct financial incentives for compliance with safeguards; 2) financiers are motivated to define and implement safeguards due to legal and political liabilities for the adverse impacts of their investments; 3) financiers are free to withhold incentives when agreed terms and conditions are not met; and 4) contingent finance respects national sovereignty since recipients may voluntarily choose to accept or reject such finance. It is therefore not surprising that entities concerned about biodiversity lobby for REDD+ funds to target biodiversity in addition to carbon (Venter et al., 2009).

The choice of the financing instrument determines the extent and nature of criteria that can be attached to REDD+ financing. So far, the UNFCCC has not made any explicit decision on the modalities of REDD+ finance, although all parties decided in 2011 that “results-based finance provided to developing country Parties that is new, additional and predictable may come from a wide variety of sources, public and private, bilateral and multilateral, including alternative sources” and that “appropriate market-based approaches [...] to support results-based actions by developing countries” (UNFCCC, 2012b: paras. 65-66) could be developed. This disperses financial risks and responsibilities so that the articulation of safeguards requirements may continue to vary by institution.

To date the most important financing modalities that are discussed for REDD+ include:

- The payment for readiness measures, including building MRV frameworks, stakeholder consultation, national strategy development (phase 1);
- The payment for policy implementation, including governance reforms, but also programmes to address drivers of deforestation (phase 2);
- The payment-for-results at the national level measured against national reference (emissions) levels (phase 3);
- Payments for demonstration projects or ‘nested’ forest carbon projects that may include payment for emission reductions at the project level.

The last two financing options could be managed through a ‘fund-based’ REDD+ system, but also through a link to carbon markets. Whatever the mechanisms for REDD+

finance, the exclusive focus of REDD+ payments on carbon has spurred fears that it will motivate the prioritisation of carbon over other values. It has therefore been suggested that the modalities of REDD+ finance could separate payments for biodiversity from payments for reduced emissions, making use of non-carbon financing (Grainger et al., 2010). The objective would be to expand the pool of potential funding sources (Bekessy and Wintle, 2008; Ebeling and Fehse, 2009; Harvey et al., 2010).

Separate biodiversity payments would, however, pose an additional burden for REDD+, requiring separate rules on design, impact assessment and payments. It will also be difficult to single biodiversity out among the many additional social, policy and environmental benefits that REDD+ should yield. It may be more feasible to consider biodiversity within the criteria that define eligibility for results-based payments (phase 3). The consideration of biodiversity and other co-benefits of REDD+ could be made a condition for multilateral and bilateral REDD+ funding. Payments could be linked to the compliance of REDD+ implementation with national planning decisions, including broader environmental outcomes. Where the conditions of finance follow a national prioritisation of area and *habitat* protection, it strengthens rather than challenges national sovereignty. As discussed in Section 5.3, this may be attractive to governments but may generate conflict with local communities and internationally-agreed objectives.

The allocation of finance to forest conservation is of particular relevance for ‘high forest, low deforestation’ countries. The definition of rules that ensure that such countries are eligible for REDD+ and the inclusion of protected areas in REDD+ could facilitate long-term gains for both mitigation and conservation by preventing deforestation from being displaced into areas that are not currently threatened (Harvey et al., 2010). Again, however, international and national demands for protected areas may be viewed as conflicting with local livelihood production and local autonomy (also see Chapter 4).

In the absence of authoritative decisions about finance under the UNFCCC, multilateral funding programmes such as UN-REDD and the FCPF have developed *de facto* methodologies for integrating biodiversity and other safeguards into REDD+ readiness (phases 1 and 2). Potentially, such programmes could provide preferential funding for multi-benefit policies (phase 2) and help in the development of systems to monitor biodiversity impacts alongside carbon MRV systems (also see Chapter 3, Section 3.4).

UN-REDD adopted its Social and Environmental Principles and Criteria (SEPC) in 2012 (UN-REDD, 2012). Grounded in international treaties, conventions and best practice guidance within the broader UN system, the SEPC are meant to assist in the evaluation of potential social and environmental impacts of national REDD+ strategies and to support countries in putting the UNFCCC safeguards into practice. The SEPC framework consists of a minimum standard risk assessment and mitigation framework, and an assessment of impact magnitude. The minimum standards ensure that the implementation of

REDD+ does not lead to social or environmental harm. The assessment of impact magnitude aims at providing guidance for designing, implementing and managing REDD+ programmes in a way that minimises social and environmental risks, and maximises multiple benefits for climate, sustainable development and conservation (Moss and Nussbaum, 2011). It is still unclear how, and under what authority, UN-REDD will monitor and enforce compliance with these standards.

The FCPF greatly exceeds UN-REDD in terms of the total pledged and potentially available funds for REDD+. The World Bank is a financially powerful actor linked with politically accountable governments, and prior to REDD+ had developed its own safeguarding system in response to past international controversies over major Bank projects (McDermott et al., 2012; World Bank, 2011). Its safeguards are designed to avoid, mitigate, or minimise adverse environmental and social impacts of all Bank projects, and are accompanied by monitoring and enforcement systems. The Bank will supervise the continued compliance of Bank-financed REDD+ readiness activities with its safeguard policies throughout the FCPF process.

In addition, countries participating in the FCPF are required to complete a ‘Strategic Environmental and Social Assessment’ (SESA). The SESA allows for the incorporation of environmental and social concerns into the national REDD+ strategy process and ensures that the FCPF readiness activities comply with World Bank policies during the strategic planning phase. One output of the SESA is the development of an ‘Environmental and Social Management Framework’ (ESMF) for managing and mitigating the potential environmental and social impacts and risks related to policy changes, investments and carbon finance transactions in the context of future REDD+ implementation. The ESMF will establish principles and criteria for policy and programme design, investment selection and, ultimately, management plans. The application of the SESA does not pre-empt the application of Bank safeguards and procedures on Bank-financed REDD+ activities in the future.

Concerns about diverging safeguards and sustainability requirements among UN-REDD, the FCPF and their financial partners have spurred the development of a ‘Common Approach’, which is meant to ensure that the various actors implementing these programmes use the same set of safeguards (FCPF, 2011). In addition, the FCPF and UN-REDD are working together on guidelines for stakeholder engagement (FCPF and UN-REDD, 2011). Still, there are important differences. McDermott et al. (2012) note that the safeguards under the FCPF can be characterised as ‘risk-based’, emphasising economic valuation of risks to minimise costs, whereas UN-REDD’s safeguard policies are more ‘rights-based’, focused on the rights of local and indigenous communities but lacking in mechanisms for monitoring and enforcement.

In contrast to the multilateral UN-REDD and FCPF, the significant quantities of bilateral aid, thus far critical for REDD+, lack the same degree of institutional standardisation. While this reduces their global transparency, it

also facilitates faster and more flexible flows of finance, highlighting tensions between international standardisation and more informal and rapidly adaptable approaches.

5.2.3.3 Market-based and hybrid governance

As discussed above, there has been considerable international resistance to the inclusion of most REDD+ activities in existing regulated carbon markets. Under the UNFCCC, only afforestation and reforestation activities via the CDM are eligible. Meanwhile the largest regulated market, the EU emissions trading scheme, has excluded credits from forestry projects (European Commission 2004: art. 11a, para. 3b). Nevertheless, smaller national and sub-national markets are developing, such as the new cap and trade programme under the California Air Resources Board. The California programme includes plans to allow the use of REDD+ credits to offset emissions, and is working with other sub-national governments in several countries on accompanying environmental and social requirements (Diaz et al., 2011).

While regulated trade in REDD+ credits is nascent and its future uncertain, a relatively small but growing voluntary market for forest carbon has developed as a form of *payment for ecosystem services (PES)* (Diaz et al., 2011). Within these voluntary carbon markets, certification is playing an increasing role as a ‘fast-track’ approach to setting standards for forest management and biodiversity.

Certification

The sale of REDD+ credits in voluntary markets sparked private efforts to develop environmental and social standards for REDD+, much in the same way certification schemes in the timber sector developed the first international standards for sustainable forest management. There has been a proliferation of such standards, with two emerging as market leaders: the corporate-driven Verified Carbon Standards (VCS) that focus exclusively on verifying saleable emissions credits, and the NGO-driven Climate, Community and Biodiversity Alliance (CCBA) that focuses on biodiversity and social co-benefits (Diaz et al., 2011). Unlike forest certification which has been limited in its market penetration, the vast majority of REDD+ carbon credits sold recently have been certified to environmental and social standards (Diaz et al., 2011), suggesting certification is becoming a necessity for market access. Thus, in the case of REDD+ project-level activities, certification provides an important mechanism for integrating social and biodiversity objectives that already carry significant market authority (Merger et al., 2011).

Hybrid (public/private) standards

REDD+ projects currently cover only a minute fraction of the tropical forest areas and lack the full scalability necessary for national-level REDD+ under the UNFCCC. Non-governmental actors including the CCBA and CARE International have therefore spearheaded a national-scale

standard-setting effort known as the REDD+ Social and Environmental Standards (REDD+ SES). The REDD+ SES is a multi-sectoral approach to allow countries to design national-level REDD+ programmes that generate significant social and environmental co-benefits.⁵ The standards are developed and tested in close cooperation with several national and sub-national governments that have volunteered to implement and test the REDD+ SES. Currently, the standards are piloted in Brazil (State of Acre), Ecuador, Indonesia (Central Kalimantan), Nepal and Tanzania.

The standards go beyond safeguarding against harm to provide a comprehensive framework to assist countries to design, implement, and assess the social and environmental aspects of their REDD+ programme, supporting and complementing the requirements of mandatory safeguards. The REDD+ SES consists of principles, criteria and indicators, and a process of monitoring, reporting and verification through multi-stakeholder assessments. A set of principles provide the key objectives that define high social and environmental performance of REDD+ programmes. One of the principles stipulates that REDD+ programmes should maintain and enhance, among others, biodiversity and *ecosystem services* (Moss and Nussbaum, 2011). The REDD+ SES standards are notable in their high level of prescription and strong emphasis on local rights and benefits. However, as with UN-REDD, it is still unclear by what authority and what mechanisms they would be monitored and enforced. This has led some researchers to hypothesise an inverse relationship between the environmental and social stringency of safeguard requirements and the accountability for enforcing them (McDermott et al., 2012). Further research will be required to assess how this varying balance between stringency and accountability affects performance, i.e. will lower standards with formal enforcement mechanisms outperform higher standards without such mechanisms, or vice versa?

5.3 National and local governance

The previous sections highlighted the tension at the international level between global governance—via intergovernmental (normative/legal), finance-based and market-based processes – and national sovereignty. This section examines within-country dynamics that stress tensions between sovereignty and local autonomy, and considers how international influence affects this balance. Many policy documents on REDD+ emphasise the need for formalising land tenure and for adopting rational systems for planning and monitoring. Such developments could facilitate international investments (e.g. Eliasch, 2008; Vatn and Vedeld, 2011), and indeed resemble long-heralded strategies for international development (Easterly, 2009). However, it is important to consider the historical and political contexts of the REDD+ countries, for whom REDD+ may appear as yet another attempt at foreign

⁵ See for more detail: <http://www.redd-standards.org/> [Accessed on: 2 July 2012]



Participation in social impact monitoring, REDD+ Pilot project, Kilwa, Tanzania; Photo © Adrian Martin

control and the favouring of Western science and rationalism over traditional knowledge and governance (e.g. Scott, 1998).

This section begins with a brief historical overview of trends in national forest governance and their intersection with REDD+, illustrated by case study boxes from the Democratic Republic of the Congo (DRC), Indonesia, Nepal and Brazil. This is followed by a summary of key lessons to be learned from this overview. It then

concludes with a review of options identified in the literature to synergise climate, biodiversity and the sustainable management of forests through national instruments.

5.3.1 National and local governance, and intersections with REDD+

Over the last few centuries and until recently, the trend in many developing countries has been towards consolidation and centralisation of state control over forest resources, initially by colonial governments and later by newly independent states aiming to strengthen their claims over the forest frontier and promote economic development (Scott, 1998). In keeping with this vision, many governments have recognised the clearing of forests as a means to claim land rights, have sponsored resettlement programmes that transplant farmers into remote forested areas, and have granted large-scale concessions for timber, mining and other extractive industries. Resources have been limited and politics contentious, preventing the formalisation of land claims amidst conflicts among indigenous peoples, local settlers and extractive industries. This has fuelled tenure insecurity across much of the forest frontier contributing to poverty and the marginalisation of rural populations (e.g. Rudel et al., 2009; Kanninen et al., 2007; Chapter 4).

The rise of international environmentalism in the mid twentieth century in many ways reinforced priorities for state control. In particular, the expansion of state-managed protected areas and national laws for species and habitat protection further alienated local populations from legal access to subsistence livelihoods (e.g. Hughes, 2006). As highlighted in Box 5.2, such environmental policies, up to and including REDD+, have often been added on top of extractive agendas – creating conflict among government ministries and failing to achieve effective conservation.

Box 5.2

The intersection between REDD+ and land use policy in the DRC

The Democratic Republic of the Congo (DRC) lies within the Congo Basin forests, the second largest tropical forest area after Brazil. Its rich biodiversity is linked to its large land mass and to a variety of physical and climatic conditions. The country, with the support of the World Bank, undertook to implement a National Forestry and Environmental Conservation Programme, adopted before its REDD+ process. However, implementation of the programme has been hampered by the absence of a sustainable and coherent land use policy instrument, leading to numerous overlaps and conflicts between protected areas, mineral exploration, forest and agricultural concessions. Meanwhile in January 2009, the Ministry of the Environment, Nature Conservation and Tourism began work on REDD+ readiness with the support of UN-REDD and the FCPF. In attempts to synergise REDD+ with the DRC's broader conservation efforts, the pre-existing National Forestry and Environmental Conservation Programme has been partially incorporated into the country's REDD+ strategy. However even if REDD+ succeeds in bringing increased funding for the DRC's conservation programmes, positive effects on biodiversity conservation are likely to be minimal without an accompanying transformation of the DRC's land tenure systems. The granting of mining concessions provides a clear case in point. According to Mertens and Belanger (2010), the Mining Register recorded 5,729 mining permits, including 463 operation permits in 2008. Some of these mines are located inside or on the periphery of classified forests. The total area covered by permits is 98 million hectares, representing 42 percent of the territory, and this area appears to be growing. For example, permission has recently been granted for hydrocarbon exploration inside the Virunga National Park. Furthermore, large-scale agricultural and road construction projects are linked to this trend. China and the DRC have signed a contract worth more than USD 9 billion for the construction of roads and railways infrastructures for mining extraction (Putzel and Kabuyaya, 2011). While in theory such expansion of infrastructure could be done in an environmentally sensitive way, there is a clear absence of an inter-ministerial coordination platform in DRC to minimise land tenure and land use conflicts across the agriculture, forest, mining, and public works sectors. Unless these conflicts are resolved in advance, the national REDD+ implementation process may increase the chaos and accelerate biodiversity loss.

While attempts to assert national control over environmental conservation have thus been frustrated by conflicting interests and inter-ministerial conflict, an increasing number of governments over the past few decades have begun to pursue decentralisation programmes (Colfer and Capistrano, 2005; Phelps et al, 2010). A host of factors has driven this trend, including the fall of authoritarian regimes, increasing national debt and structural adjustments (curtailing resettlement programmes and cutting government budgets), and growing awareness of the potential for community-driven resource management to deliver both social and environmental benefits (see Chapter 4). In some countries, such as Nepal, Mexico and Papua New Guinea, decentralisation has involved an extensive handing over of management, resource and/or land rights to local communities. It is now estimated that roughly 22 percent of developing country forest area is under some degree of community control (Molnar et al., 2010). This trend has also affected government approaches to protected areas, leading to the designation of ‘community protected areas’ particularly in the buffer zones of national parks. The case study from Indonesia provides a positive example of such an approach, which has since received support as a ‘REDD’ project (Box 5.4).

The analysis of Meru Betiri may highlight the potential of community participation in protected areas management to produce optimal ‘win-win’ solutions for REDD+, biodiversity and other co-benefits. Meanwhile, in Indonesia and elsewhere the expansion of protected areas is being put forward as the core national level strategy for integrating biodiversity and REDD+. At the national level in Indonesia, such strategies appear to be accompanied by centralised policy-making and target-setting, supported and encouraged by international REDD+ donors (see Box 5.3).

Indonesia’s national REDD+ strategy

Box
5.3

Indonesia drew massive attention when its President committed, conditional upon international support, to reduce the country’s greenhouse gas emissions by 41 percent by 2020, and in response, the Government of Norway committed USD 1 billion to support REDD+ in Indonesia. A national strategy is currently undergoing final drafting by the REDD+ Task Force, and has included biodiversity issues as a priority. For example, “the improvement of the sustainability of biodiversity” is stated to be part of the scope of REDD+ activities (REDD+ Task Force, 2012). The strategy goes further by stressing that forests which have a high concentration of carbon and biodiversity will become protected areas, with strong emphasis on the improvement of forest governance for REDD+ and the synergy between different types of laws which aim to conserve biodiversity, forests and natural resources, and to regulate their exploration, development and exploitation.

The case of Nepal (Box 5.5) illustrates how national commitments to expand protected areas as part of a REDD+ strategy have raised concern among some local populations that REDD+ could thereby undermine locally-driven sustainable management of forests.

While protected areas may be a favoured REDD+ biodiversity strategy for some national government actors, other governmental and non-governmental actors operating at the project level have focused efforts on market-based payments for ecosystem services (PES) and other economic incentive mechanisms (also see Chapter 4, Section 4.5.2). In Brazil, sub-national state governments have played a key role in spearheading such approaches,

The intersection of REDD+ and community buffer zone management in Indonesia

Box
5.4

Despite Indonesia’s commitment to conserving its biological resources through the establishment of national parks, during the reform period of the late 1990s-early 2000s, rates of deforestation inside Meru Betiri National Park were unprecedented (Casson et al., 2006). The park lost approximately 2,500 hectares of forest during this period as companies and small-scale farmers competed for remaining forestland. However, as the park’s biodiversity came under threat, an interesting experiment in the buffer areas of the forest provided valuable lessons.

Curahnongko village is located in the buffer zone of Meru Betiri National Park and beginning in 1994, Lembaga Alam Tropika Indonesia (LATIN), the Forest Department of Bogor Agriculture University and the villagers of Curahnongko established and maintained a seven-hectare demonstration plot to cultivate medicinal plants and promote agroforestry practices (Aliadi, 2010). While other parts of the park were being devastated, the community-managed demonstration site remained intact. In an effort to stem rates of deforestation, Meru Betiri Park authorities approached LATIN to replicate the demonstration sites with reforestation activities on plots throughout the park. In 2001, 3,500 households from five villages (Curahnongko, Andongrejo, Sanenrejo, Wonoasri and Curahtakir) were recruited to participate in a forest rehabilitation programme. By 2004, some 2,250 hectares of land that had previously been encroached upon had undergone reforestation efforts. In total, 104 community forestry-farmer groups in cooperation with local NGOs were responsible for initiating the planting of 23,027 seedlings (Aliadi, 2010).

While the communities remained without formal rights to the forestland, the livelihood benefits they were able to secure through agroforestry and the cultivation of medicinal plants amongst others were sufficient to incentivise them to play a critical role as forest stewards. The relationship between park authorities, local communities and supporting NGOs has evolved such that in 2010, the ‘Meru Betiri National Park – Reducing Emissions for Deforestation and Degradation+’ (MBNP-REDD+) pilot project was launched in 58,000 hectares of Meru Betiri National Park, including 4,000 hectares of the ‘rehabilitation’ lands under the management of the local communities (ITTO, 2010).

Protected areas and REDD+ in Nepal

In Nepal, the principal approach to biodiversity conservation and related governance is protected areas. Of the country's 20 protected areas (Khatri, 2010), 16 parks are under government management. In terms of the geographical area coverage, 62 percent of all protected areas are co-managed with support from the local communities living in and around them. However, protected area-based conservation approaches have drawn criticism due to their failure to secure effective participation of dependent communities in their planning and management. Lack of effective consultations during their establishment, including with respect to FPIC, and unclear tenure rights for the local communities living in the buffer zones have raised questions on the rhetoric and reality of participatory protected area management in the country (Budhathoki, 2011).

Further, in the context of REDD+, the current approach of protected area-based biodiversity conservation is seen by many, including by community forest user groups, as an approach to reconsolidate control over previously devolved forests. Such concerns have been shared by local stakeholders and civil society organisations during the implementation of the grassroots capacity building for REDD+ projects in Nepal by RECOFTC (Regional Community Forestry Training Centre) in partnership with FECOFUN (Federation of Community Forest Users, Nepal - Bhandari et al., 2012). To some extent, such concerns are also based on the substantial increase in the coverage of protected areas over the years. While in 1975, protected areas covered 4,376 km² of the country's forests, currently, this network has grown to a total of 34,186 km² of forest area, about 23 percent of the total territory of Nepal. These developments are interpreted as renewed interest by the government in national forests (Bushley and Khatri, 2011). By monetising forest carbon, the market value of forests, including those previously considered marginal, may further incentivise the central government to increase control over forest lands.

REDD+, biodiversity conservation and forest management in Brazil

Acre's State System of Incentives for Environmental Services (SISA) was initiated by the state government and passed into law in 2010 (Law 2308/2010). The system focuses on the conservation and recuperation of seven environmental services: 1) carbon sequestration and enhancement of stocks through forest conservation and management; 2) natural scenic beauty; 3) socio-biodiversity; 4) water and hydrological services; 5) climate regulation; 6) appreciation of cultures and traditional ecological knowledge; and 7) conservation and recuperation of soils (Government of Acre, 2010). The SISA is based on Acre's policy for the valuation of environmental assets, which involves recuperation of degraded lands (through reforestation and revitalised agricultural production) and valuation of standing forests (through forest management, certification of sustainable rural properties and payments for environmental services). It is the first state law to highlight the provision of a variety of environmental services, including biodiversity. One specific biodiversity conservation strategy included in SISA is the planned creation of protected areas along the BR-364 highway to buffer against the negative impacts of imminent further road development. This action is based on lessons learned from past deforestation in the eastern part of the state, where many municipalities have more than 50 percent of their area deforested (Salimon and Brown, 2009). Biodiversity monitoring in SISA will likely be facilitated through the use of the extensive Rainfor permanent plot network already in place and through the close relationship between environmental researchers and decision-makers in Acre.

Another example of state-level innovation is *Cotriguaçu Sempre Verde* in northwest Mato Grosso, which is led by the 'Instituto Centro de Vida', The Nature Conservancy, an affiliate of the National French Forest Service (ONF-Brazil) and the state environmental secretariat. The forest sector in the municipality of Cotriguaçu is dominated by the existence of perverse incentives that encourage illegal logging due to difficulties that producers face in obtaining official harvest permits (IFT and ICV, 2010). To address this challenge, project proponents entered into collaboration with the 'Instituto Floresta Tropical' to create PRODEMFLOR (Forest Management Development Programme) in the REDD+ project area. The goal of PRODEMFLOR is to promote *reduced impact logging* in Cotriguaçu through voluntary, written agreements with small to medium-sized timber companies. Timber producers who sign onto PRODEMFLOR are required not only to improve their forest management practices, but also to commit to increased transparency in their operations. In exchange, the companies receive training in forest management and support in applying for official harvest permits. Under the PRODEMFLOR umbrella, proponents provide reports from remote sensing analyses and field assessments associated with specific forest management plans to highlight aspects that would aid or impede the companies in obtaining harvest licences. All costs of the pilot phase of PRODEMFLOR are subsidised by external project donors with the idea that timber companies will eventually cover these costs to acquire harvest licences more easily. If successful, PRODEMFLOR has the potential to expand to other Amazonian municipalities and evolve into a system that will track and attest to the sound origin of timber for the regional industry to encourage forest conservation through sustainable timber production.

suggesting their attractiveness as a means to capture benefits at a local scale (Box 5.6).

5.3.2 Conclusions from national and local analyses

Several inferences can be drawn from the above analyses. First, the framing of REDD+ as a national-level incentive system under the UNFCCC, while necessary to gain the support of Parties to the Convention, has generated local concerns about recentralisation and the loss of local livelihoods and autonomy. The multivalent, fragmented and inconclusive nature of the international REDD+ complex has created space for considerable local innovation but does little to ensure desired environmental and social outcomes. Meanwhile there is substantial risk that the assertion of international authority through REDD+ finance could redirect attention away from previously successful non-REDD+ activities and worsen existing social imbalances and conflict.

The Brazil and Nepal cases illustrate how some countries have been active and effective in promoting forest conservation through efforts pre-dating and/or largely independent of international REDD+ funding. This finding suggests that ‘country-driven’ efforts, as emphasised within the UNFCCC, are crucial. Likewise, several of the case studies emphasise the importance of local, community-level engagement and buy-in. As is evident from the Nepal case study (see Box 5.5), national and local objectives do not always match, highlighting the challenges inherent in reaching an aspirational goal of widespread, multi-scale acceptance of REDD+.

Actions labelled explicitly as REDD+ form just one small part of a larger forest and biodiversity governance complex (e.g. land tenure regimes, community-based governance, national park systems). Rather than begin with the question of how to make REDD+ work for biodiversity, the question might be better framed as how to achieve the sustainable management of forests and biodiversity conservation more broadly – whether through REDD+ or other means in a manner that is socially and politically informed.

5.3.3 Options to synergise climate, forest management and biodiversity objectives through national instruments, and their intersection with local forest governance

In light of the above analysis of how REDD+ is currently unfolding at national and local levels, this section critically reviews the existing literature on ways to improve the incorporation of forest management and biodiversity objectives into national REDD+ strategies and measures (see also Annex A for a brief overview of the opportunities and risks of such an incorporation). The instruments

are divided into data collection and information gathering, policy, regulatory and finance (incentive) measures.

As discussed in Chapter 4 on REDD+ ‘trade-offs’, the governance of carbon, forest management and biodiversity has profound implications for local social welfare. While an assessment of social safeguards is beyond the scope of this chapter, the following analysis highlights how particular policy approaches differently empower global, national or local actors in REDD+ decision-making.

5.3.3.1 Information and data collection

Creating approaches to systematically gather and report data on the impacts of REDD+ actions can inform a country’s REDD+ strategy design, guide investments to specific areas and activities that maximise benefits, and ensure that actions taken are not harming people, ecosystems and wildlife (Lee et al., 2011). In order to understand long-term effects, information collection must be repeated and a system of continuous monitoring as well as a periodic review of a country’s REDD+ policies put in place. Considering the existing obligations to collect and report information under various international agreements and programmes, options for leveraging existing data and systems include (adapted and expanded from Lee et al., 2011):

- Building on forest inventory reporting. For example, considering additional indicators to forest inventories, such as number of plant/animal species and the extent of ecological networks, to ensure that REDD+ actions also deliver co-benefits.
- Using remote sensing data that is aimed at assessing carbon stock changes for monitoring of multiple benefits; this will also help to ensure consistency of data sets used.
- Using existing data sets, for example on soils, run-off and precipitation, to assess the effects of forest protection or reforestation on a watershed.
- Creating indicators for socio-economic benefits of REDD+ activities that build on national monitoring of socio-economic statistics. For example, Peru is considering possible indicators, such as: jobs created, family income statistics and food security for forest dwellers.⁶
- Systematically collecting information generated by voluntary carbon projects, environmental impact assessments, and other privately-collected information.
- Centrally collecting, analysing and storing information gathered under multilateral agreements and regional programmes, such as the CBD, UNFCCC and the Ramsar Convention on Wetlands. For example, NBSAPs or criteria for SFM contain elements relevant for biodiversity and REDD+ (CBD, 2012).
- Taking into account the scarcity of data and lack of capacities in many countries, Gardner et al. (2012) have proposed a tiered approach to biodiversity monitoring that is partially analogous to the Intergovernmental

⁶ Readiness Preparation Proposal (R-PP) submission to the FCPF

Panel on Climate Change's guidance on tiered-emissions reporting, in which lower tiers can provide a realistic starting point for countries with fewer data and lower technical capacities.

- Community-based monitoring of carbon, forest management, biodiversity, etc. (Fry, 2011).

The last of the above recommendations refers to approaches particularly well suited to engaging local communities, potentially involving methodologies that contribute to their understanding and empowerment. As illustrated by the case study from Nepal (Box 5.5), there is otherwise a risk that REDD+ as a mechanism will be applicable only to those with technical or scientific understanding, thereby losing the substantial knowledge, as well as buy-in, of local communities some of which have served as effective long-term forest stewards.

5.3.3.2 Planning and strategy

Data on forest management and biodiversity, if gathered in a manner meaningful both to policy-makers and to their stakeholders, can inform interested parties about the potential trade-offs and synergies of pursuing particular REDD+ strategies and/or broader low-carbon development strategies. Rather than creating entirely new monitoring systems, their integration into existing planning tools may help to reduce the overall costs and build a more coherent REDD+ policy framework.

Spatial analysis allows the identification of areas of high ecological value and biodiversity, potential leakage areas and areas of important ecological connectivity (CBD, 2011). It also helps identify gaps in existing networks of protected areas, of ecosystems and habitats that are under-represented and require particular attention and protection (Paoli et al., 2010). Such gap analyses can inform decisions about classification or re-classification of forested land, including the cancellation of concessions and re-classification of land to forbid conversion, the restricting of forest management practices, and the extension of a network of protected areas.

However, the decision of how to prioritise forest management and biodiversity objectives relative to other values, such as local livelihoods or economic production, is ultimately a political one. Evolving international principles of 'good governance' (as articulated e.g. in the Aarhus Convention) emphasise the need for broad-based participation in determining priorities for land use, and in designing socially acceptable means to achieve them. Otherwise plans for habitat and species conservation may fail to be implemented, as illustrated in Box 5.2 on tenure conflicts in the DRC.

5.3.3.3 Policies and measures

Based on the broad directions formulated in REDD+ strategy and planning documents, governments can take various measures to sustainably manage forests and protect biodiversity in the context of REDD+, as well as to increase effective coherence and consistency among

measures aiming at forest management, mitigation and *adaptation* in the land use and forest sector, and biodiversity protection. In addition, following is a (non-exhaustive) list of policy options to ensure biodiversity protection in the context of REDD+:

- Legal reform: The clarification of land tenure, land use and relevant rights (to forests, carbon, biodiversity) (Swan and McNally, 2011); improved legal coherence across forest, mining and agricultural sectors; and improved enforcement of existing laws may be more important to sustainable forestry and biodiversity conservation than new policies.
- Community management: Strengthening the legal framework for customary forestland tenure and management practices can empower local communities as effective stewards of forest carbon stocks and biological diversity in the longer term (Swan and McNally, 2011).
- PES: The development of legal frameworks to govern payment for ecosystem service schemes could increase the market value of these services while simultaneously addressing biodiversity and social welfare (Greiber, 2009).
- The adoption of explicit national targets for ecosystem and species protection across the full range of native ecosystem types and biogeographic sub-regions (Paoli et al., 2010).
- The use of context-appropriate strategies to incentivise conservation in areas with high forest cover and low deforestation rates, in particular if they have high biodiversity value (Harvey et al., 2010).
- Within forests of identical carbon stock, the prioritisation of REDD+ implementation in those of greatest biodiversity value and which contribute most to landscape connectivity (Harvey et al., 2010).
- The establishment of protected areas is usually motivated by ecological concerns (as well as, in some cases, social concerns), and they are therefore also likely to provide non-carbon benefits (Lee et al., 2011). Natural forest carbon stock enhancement activities under REDD+ could also promote broad-scale *forest landscape restoration*, thus significantly expanding forest quality and quantity across the tropics (Swan and McNally, 2011).
- Governments may also directly forbid or mandate certain actions, including particular forest management practices (Swan and McNally, 2011). Investors and other entities that engage in specific REDD+ programmes or projects can be held accountable for the impact of their activity through strategic environmental assessments (SEAs) and environmental impact assessments (EIAs). They would also have to comply with relevant safeguards, which mandate no-harm as well as taking action to maximise benefits.

All but the last of these identified options emphasise national-level authority and scientific and 'technical' assessment, from target-setting, to rational land use zoning, to enhanced regulation and enforcement. In light of the analysis in Section 5.3.1, such approaches may in some

cases prove conflictive and risk contradicting effective locally-driven solutions. This highlights the importance of holistic thinking that integrates biodiversity goals within a broader framework of good governance.

5.3.3.4 Finance and incentives

Countries could also structure particular incentives to ensure the protection of biodiversity in addition to REDD+:

- A single payment system that combines carbon and biodiversity benefits. Countries could adopt a PES system that involves financial arrangements with private landholders or communities to protect ecosystem services. Such a PES system has the added benefit of valuing ecosystems and compatibility with participatory forest management and can provide an alternative to, or be combined with, national-scale financing systems or carbon market options (Lee et al., 2011).
- Incentivise or require biodiversity safeguards in carbon markets. Countries could support the use of FSC, CCBA or other standards to certify and market carbon offsets. The use of carbon markets can be seen as a special case of PES focusing on greenhouse gas regulation. The compliance with particular biodiversity safeguards can be included in the eligibility and approval criteria of forest carbon projects. In this case, verification of results could be part of the evaluation of a project's climate and biodiversity benefits according to the regulatory criteria and the project's monitoring plan. Surveys have also confirmed that buyers of carbon credits are willing to pay a premium for carbon credits that meet high social or environmental standards (Neeff et al., 2009).
- Adopt a separate, parallel biodiversity incentive system. Adoption of a separate payment system that gives communities, landowners, project developers, etc. (as appropriate) access to additional (non-REDD) finance in cases where they deliver biodiversity benefits in addition to emissions reductions.
- Support existing efforts already proven to generate positive biodiversity outcomes. As evident from the national and local-level analysis in this chapter as well as in Chapter 4, the best balance of environmental, social and economic objectives may sometimes be achieved without external finance or through finance that supports existing governance systems that are already achieving desired synergies.

As discussed in Section 5.3.1, sub-national governments and non-governmental actors have been instrumental innovators of PES approaches to REDD+. For example the SISA system in Acre State (Box 5.6) resembles the first approach suggested above. It is as yet unclear what the most appropriate role is for national governments in such cases - i.e. the appropriate balance of national standardisation and legalisation, and flexibility for sub-national and voluntary innovation. Meanwhile some non-governmental stakeholders, as illustrated in Box 5.5 on Nepal, are concerned that the monetisation of forest values will lead to the alienation of forest resources from local



FSC certified Mpingo (*Dalbergia melanoxylon*), Kilwa, Tanzania.
Photo © Adrian Martin

and subsistence users. Hence in some cases the greatest synergies may be achieved through no action, and/or finance expressly designed to support existing governance systems. Appropriate financing of REDD+ requires more than the funding of new institutions, policies and incentive schemes, but rather the careful consideration of how REDD+ finance interplays within the broader socio-political landscape.

5.4 Conclusions

This chapter has examined the emergence and evolution of REDD+ within the broader landscape of climate, forest and biodiversity governance, and the lessons this holds for developing environmentally and socially synergistic policies. A diverse institutional complex has developed to govern REDD+ that draws variously on three major sources of authority: (sovereign) governmental, fund-based and market-based. Each source offers different opportunities and constraints.

Intergovernmental negotiations have drawn on governmental authority to produce relatively widespread agreement on the singular goal of emissions reductions, but few binding commitments regarding sustainable management of forests and biodiversity conservation. These latter objectives have been addressed through broad normative guidance, commitment to monitoring and reporting, and activities such as timber legality verification that reinforce state sovereignty. The development of internationally-standardised safeguards for forest management and biodiversity has occurred primarily through fund-based and market-based initiatives. Fund-based REDD+ activities enable financial institutions to make payments contingent on compliance with their own operational standards,

relatively less constrained by market competition or the need for intergovernmental consensus. However, their impact is reduced by the limited quantity of funds available, and the limited capacity of REDD+ countries to meet diverse operational requirements and absorb funds.

Market-based approaches to REDD+ are currently restricted to voluntary markets, where certification has offered a 'fast-track' means for NGOs and other actors to develop ambitious environmental and social standards for PES projects. While many of these projects link payments only to carbon, they could extend to other ecosystem services such as biodiversity or even livelihood provision. However the small size of voluntary markets, and the proliferation and competition among certification schemes, significantly constrain their impacts. The scale of market standardisation could increase if REDD+ is included in state-based 'cap-and-trade' systems, but with unknown effects on environmental and social requirements.

Efforts to promote REDD+ safeguarding at the international level may either complement or constrain national sovereignty and local autonomy. National governments play a key role in designing and implementing

country-appropriate legal reforms, but suffer from lack of capacity and competition among ministries. International support may facilitate country-led efforts and/or heighten conflict by favouring particular ministries or actors. Likewise, international and national REDD+ efforts may empower local communities to act as stewards of biodiversity via community-based tenure arrangements, or constrain local autonomy through the expansion of strictly protected areas. Table 5.1 below summarises these findings.

Taken as a whole, it is clear that the integration of forest management, biodiversity, and social and political concerns into REDD+, has thus far involved a diverse array of institutions and policies drawing on different sources of authority. Given the power struggles and inherent trade-offs involved, REDD+ governance is likely to remain pluralistic and contested. As observed in the previous GFEP report (Rayner et al., 2011), the most effective way forward may be to better understand, embrace and engage with this complexity rather than attempt to impose singular solutions.

The potential role of different sources of authority in supporting different governance strategies for REDD+ safeguards

Table
5.1

	Governmental		Fund-based (national and project-level)	Market-based (project level)
	International	National		
Legal reforms	Trade restrictions on illegal products, public procurement policies (e.g. EU Timber Regulation, Lacey Act)	Improved legal coherence, tenure reform, tax incentives, enforcement	Operational policies/safeguards	Legality certification
Community mgmt	General normative guidance	Tenure reform/ decentralisation	Operational policies/safeguards	Timber/NTFP/ Carbon certification
Legality verification	Partnership agreements to stem illegal trade; e.g. FLEGT VPAs	Legality assurance schemes	Operational policies/safeguards	Legality certification
PES	Rules for carbon trading	(New) legal frameworks for PES	Operational policies/safeguards	Carbon/Biodiversity certification
Biodiversity/ social standards	General normative guidance	EIAs, SEAs, Biodiversity laws	Operational policies/safeguards	Timber/NTFP/ Carbon certification

Strong leverage points
 Lesser leverage points

Annex A

Opportunities and risks for biodiversity under REDD+

This annex summarises the opportunities and risks related to addressing biodiversity over the three phases. While the focus of the table is on biodiversity, the concepts also apply to other objectives associated with the sustainable management of forests.

Biodiversity and phases of REDD+ implementation		Table A.1
	Opportunities	Risks
Phase 1 Readiness	Integrate biodiversity in early planning processes and MRV systems. Build capacity to identify risks and synergies for biodiversity conservation.	Failure to consider biodiversity in the readiness phase may be hard to mitigate as this phase will establish the basic systems and tools to implement REDD+.
Phase 2 Policy Implementation	Identify policies and measures that display 'win-win' synergies. Conduct strategic assessments to avoid adverse impacts of REDD+ measures. Include biodiversity in stakeholder consultations.	Adverse effects on biodiversity of REDD+ policies and measures focused primarily on carbon. Ineffective policies. Fragile states may not be able to protect sensitive ecosystems and focus on protecting carbon-rich forests.
Phase 3 Payments-for-results	Protection of biodiversity can be a payment condition / a premium can incentivise additional measures.	Data may be incomplete or erroneous. Fragmentation of the system through multiple and conflicting donor requirements.
Demonstration projects	Demonstration projects may test results-based payments that incorporate biodiversity.	Most demonstration projects are driven and implemented by private actors; coordinated regulation of these projects may be difficult.

Annex B

Guidance from the CBD for integrating biodiversity into REDD+

The protection of biodiversity defines the core mandate and objective of the Convention on Biological Diversity (CBD). While not concerned with REDD+ *per se*, the CBD seeks to ensure that biodiversity is given due consideration in the implementation of international and national policies. The CBD also re-groups (or 'bundles') knowledge and expertise around biodiversity impact and monitoring. In the context of REDD+, the CBD can inform biodiversity safeguards and formulate indicators for the design, implementation and continuous monitoring of REDD+.

As concrete guidance for parties involved in REDD+, the second CBD Ad Hoc Technical Expert Group (AHTEG) on biodiversity and climate change developed basic recommendations to support Parties in their efforts to implement REDD+ in a way that is supportive of CBD provisions. The AHTEG recommendations led to a number of decisions at the tenth CBD Conference of the Parties (COP), which provide, among others, a mandate to the Executive Secretary (without preempting future decisions of the UNFCCC) to provide advice on appropriate safeguards for biodiversity; identify possible indicators to assess the impacts of REDD+ on biodiversity and assess potential mechanisms to monitor impacts on biodiversity from these and other ecosystem-based approaches for climate change mitigation measures.

Based on the results of the AHTEG, the CBD COP adopted guidance on ways to conserve, sustainably use and restore biodiversity and ecosystem services while contributing to climate change mitigation and adaptation, thus supporting the implementation of REDD+ safeguards (Decision X/33, paragraph 8). This guidance refers to, among others: the implementation, as appropriate, of improved land management, reforestation and *forest restoration* prioritising the use of native communities of species, to improve biodiversity conservation and associated services while sequestering carbon, and limiting the *degradation* and clearing of native *primary* and *secondary forests*; the execution of strategic environmental assessments (SEAs) and environmental impact assessments (EIAs) that facilitate the consideration of all available climate change mitigation and adaptation options; or the consideration of incentives to facilitate climate change related activities that take into consideration biodiversity and related social and cultural aspects (Decision X/33).

The CBD could also complement REDD+ safeguards, in particular where they fall short of considering particular biodiversity risks, such as the risk of afforestation in areas of high biodiversity value. The guidance on afforestation, reforestation and forest restoration provided by the CBD in paragraph 8(p) of Decision X/33 could fill this gap, to cover the possibility that activities considered as part of 'enhancement of forest carbon stocks' under REDD+ serve to reduce biodiversity (CBD, 2011). Similarly the risks of displacement of deforestation and forest degradation to areas of lower carbon value and

high biodiversity value are not adequately covered under the emerging UNFCCC rules, and it would be helpful to consider the ecosystem approach in this context (CBD, 2011).

Of further relevance is the CBD's Expanded Programme of Work on Forest Biological Diversity (Decision VI/22, annex) and the new Strategic Plan for the Convention for the period 2011 to 2020 (Decision X/2). The Expanded Programme of Work consists of 130 measures that are to be implemented in accordance with national priorities. Relevant measures include those that control forest fires, improve forest governance and promote sustainable management of forests. The success of REDD+ will also determine the feasibility of the targets formulated in the Strategic Plan (CBD, 2011).

When it comes to concrete monitoring and data collection, the CBD Secretariat has also been requested, in Decisions IX/5 and X/36, to further enhance report streamlining based on the Collaborative Partnership on Forests' (CPF) Task Force on Streamlining Forest-related Reporting, and to investigate whether there are inadequacies in forest biodiversity reporting and monitoring, with the objective of further improving the biodiversity component of the Global Forest Resources Assessment and other relevant processes and initiatives. The CBD also has a programme of work on protected areas (POWPA - Decision VII/28) that contains multiple objectives with time-bound targets. As part of this work programme, parties were guided to execute a gap analysis of their protected area system by the end of 2006. The results of this exercise, as well as other elements of the POWPA, are relevant in the context of REDD+ planning.

The CBD complements the UNFCCC also in defining indicators for biodiversity assessments, and social and environmental impact evaluation. Biodiversity impacts and impacts on indigenous and local communities due to REDD+ activities should be compared against the most likely scenario in the absence of REDD+ activities (CBD, 2012). Pursuant to Decision X/33 paragraph 9 (h), proposed indicators for the possible monitoring of the contributions of REDD+ to the objectives of the CBD are understood to comprise impacts on biodiversity, and on the traditional knowledge and customary sustainable use of indigenous and local communities (Articles 8(j) and 10(c) of the Convention). In 2012 the CBD's Executive Secretary proposed a number of biodiversity and policy indicators (describing on the one hand, the state of biodiversity and ecosystems, and on the other, providing information on the full and effective participation of indigenous and local communities and the involvement of biodiversity experts) (CBD, 2012). The indicators are divided into global indicators ready to be implemented, global indicators that need further elaboration, and national and other sub-global indicators.

Annex C: Regional governance in REDD+ and FLEGT processes: the case of COMIFAC

The Central African Forests Commission (COMIFAC) is a regional organisation of the ten states in the forests of the Congo Basin. The groundwork for COMIFAC was laid in the 1999 Yaoundé Declaration by the Central African Heads of State for the 'Conservation of the Congo Basin' and formalised in the 2005 Brazzaville 'Treaty on the Conservation and Sustainable Management of Forest Ecosystems in Central Africa'. COMIFAC provides an example of the opportunities and challenges of regional coordination within both REDD+ and FLEGT processes.

COMIFAC is involved in REDD+ mechanisms in two ways. Firstly, at the political level, from 2005- 2011, COMIFAC countries submitted seven requests to the Subsidiary Body for the Scientific and Technological Advice of UNFCCC. These submissions related notably to: funding sources; field of application; methodological and technical questions; reference scenarios and scale. During the Copenhagen COP (2009), COMIFAC countries underscored their need to strengthen their technical capacity for monitoring forest cover and carbon stock. This position has been recalled during the Joint Declaration of Intent on REDD + in the Congo Basin published during the Durban COP. Secondly, at the ground level, COMIFAC is administrative supervisor of two REDD+ projects: i) The 'Regional REDD Capacity Building Project' with the support from the World Bank/GEF; and ii) The new regional initiative project on REDD+ which will help ten Central African countries to set up advanced national forest monitoring systems. This latter forestry project will be managed jointly by the COMIFAC and FAO in close collaboration with the Brazilian National Institute for Space Research. The Congo Basin Forests Fund, launched by the Governments of Norway and the United Kingdom through the African Development Bank is funding the initiative with EUR 6.1 million. This project will reinforce regional capacity and allow COMIFAC countries to strengthen their cooperation in the forestry sector, in particular with regards to their capacities to provide transparent and reliable data and information on forests.

Another example of COMIFAC involvement in global forest processes is the FLEGT support project for the six timber producing countries of the Congo Basin, implemented under COMIFAC with the financial support of the European Union. To date, FLEGT has focused primarily on Voluntary Partnership Agreements with individual countries, but greater participation of COMIFAC may be critical for providing accurate data on transboundary trade flows and the related traceability of the timber exchange.

In sum, COMIFAC as a regional coordinating body could play an important role in both REDD+ and FLEGT processes, but it remains to be seen how such coordination will work in practice. There is much untapped potential for regional intergovernmental actions to support global initiatives on biodiversity conservation, forest management and REDD+ and their national ownership.

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Chapter 6

Conclusions

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6.1 REDD+: Opportunity and risk

Forests, especially those in tropical and sub-tropical regions, contain most of the world's terrestrial biodiversity and provide a broad range of ecosystem services. These services directly benefit people both globally and locally, in particular the hundreds of millions of people whose livelihoods depend, at least in part, on forests. One of these global services - carbon sequestration - is receiving international attention because of forests' important contribution to the global carbon cycle.

Deforestation, resulting mainly from ongoing conversion of forests to agricultural land, is the major cause of global biodiversity loss in terrestrial ecosystems. It is also the second largest anthropogenic source of carbon dioxide emissions to the atmosphere after fossil fuel emissions. Forest degradation (changes in forest condition that affect a forest's capacity to provide goods and services) is a major contributor to global anthropogenic CO₂ emissions, and an important driver of biodiversity loss.

The future of the global forest carbon sink – as well as the world's terrestrial biodiversity - is highly uncertain. On the one hand, ongoing threats from land-use and environmental change are significant, and on the other, there are potentially significant opportunities for positive change through efforts to reduce rates of deforestation and forest degradation. The Intergovernmental Panel on Climate Change (IPCC) has highlighted that reducing deforestation, especially in the tropics, can considerably reduce greenhouse gas emissions and increase CO₂ removals at low costs, and can be designed to create synergies with adaptation and sustainable development.

REDD+ activities aim to mitigate climate change by reducing the greenhouse gas emissions resulting from deforestation and forest degradation. A number of actions, including changes in land use and management practices (in both forested and non-forested land) can achieve REDD+ objectives while also conserving biodiversity and enhancing the provision of other forest ecosystem services. Selecting the most appropriate approaches for

implementing such actions is critical to ensuring the best outcomes for biodiversity, carbon, and other ecosystem service benefits, and for people. Importantly, given the complexities of forest ecosystems and their management, and their importance for biodiversity and human well-being, poorly designed and implemented REDD+ interventions could have serious adverse impacts on biodiversity and people.

For these reasons, a thorough understanding of the relationship between biodiversity, carbon and other services in the context of the ecology of forests and multiple-use landscapes, and of the impacts of human activities on these relationships, is essential to inform appropriate management actions. It is also of crucial importance that any intervention be considered within the governance context and related constraints of a given region.

In this report, we have synthesised and analysed current knowledge regarding: the relationships between forests, biodiversity and carbon, and other ecosystem services; how these complex relationships may be affected by deforestation, forest degradation, and the management activities implemented to achieve REDD+ objectives¹; the potential synergies and trade-offs between and among environmental and socio-economic objectives, and; their relationship to governance at multiple scales.

6.2 Relationships between forest biodiversity, carbon and other ecosystem services

Biodiversity is of fundamental importance to forest productivity and other critical ecosystem processes and services. While some forest ecosystem services, such as erosion control, are only weakly related to biodiversity, losses of biological diversity can adversely affect the resilience of forest ecosystems to ongoing human impacts, environmental change, and the long-term provision of many ecosystem services, including carbon storage.

Together, deforestation and forest degradation are responsible for some of the greatest negative impacts on terrestrial biota with very extensive areas affected in both tropical and sub-tropical regions. Deforestation, essentially via conversion for pasture or intensive agriculture, substantially reduces ecosystem carbon stocks, prevents recovery of carbon stocks and results in an almost total loss of a site's original biodiversity, with reduced ecosystem function. Forest degradation resulting mainly from fragmentation, human-induced fires and unsustainable forest management, can also have severe adverse effects on biodiversity and may significantly reduce carbon stocks and the provision of ecosystem services. Further deforestation may ensue, as unsustainable forest management can increase access to previously remote areas,

making the forest more susceptible to conversion for agricultural use or fires. The combined impacts of past and ongoing degradation on forest carbon and biodiversity may approach those of deforestation.

Ecological thresholds exist in ecosystems that, if crossed, can result in detrimental outcomes for ecosystem function and reduced provision of ecosystem services. To prevent the system from crossing these thresholds, forest management should strive to use goods and services at levels known to be sustainable for the ecosystem (i.e., within a 'safe operating space').

Different forest types and ages are highly variable in species richness and their capacity to store carbon, with for example primary forests storing more carbon and growing (secondary) forests sequestering carbon more rapidly. Accordingly, land use planning processes need to take these differences into account when addressing both biodiversity and carbon objectives.

Experimental results indicate that increases in tree species richness in planted forests can increase biomass carbon stocks, although at high levels of diversity in forests, the relationship between changes in species richness and carbon stock changes remains poorly understood. Given the inherent difficulties in quantifying the functional importance of all species, management of forest ecosystems should take a precautionary approach to safeguarding biodiversity.

In (sub-) tropical forests that are allowed to naturally regrow or recover from disturbance and degradation, carbon and biodiversity can both increase over time. However, the rate at which they recover diminishes over time, and recovery of biodiversity can be slower than that of carbon. Secondary forests are of significant value to conservation of both carbon and biodiversity but there is uncertainty with respect to the extent to which 'novel' forest ecosystems will be able to provide expected ecosystem goods and services.

Due to the large number of endemic species, endangered species, and unique species assemblages in (sub-) tropical forests, spatial planning for biodiversity conservation objectives needs to be more area-specific than is necessary for carbon management.

6.3 Impacts of management actions

Implementation of REDD+ activities is achieved through management actions in both forest and non-forest land. Individual actions often address more than one REDD+ activity.

Overall, 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

¹ The activities considered in this report relate to those specified in the UNFCCC's Cancún decision on REDD+ (Decision 1/CP.16 paragraph 70) that encourages developing country Parties to contribute to mitigation actions in the forest sector by undertaking activities to: reduce emissions from deforestation and forest degradation; reduce emissions from forest degradation; conserve forest carbon stocks; sustainably manage forests; and enhance forest carbon stocks.

benefits for both carbon and biodiversity compared to actions that seek to restore them. However, securing positive outcomes for both will depend on several factors such as location, scale of implementation, initial conditions, historical impacts, forest type and the wider landscape context. The timing of benefits is also likely to differ: actions to avoid deforestation and degradation can yield immediate carbon and biodiversity benefits, while those that seek to restore forests yield biodiversity benefits more slowly than carbon benefits. A consideration of this context dependency is essential in planning REDD+ actions across different sites and into the future.

Importantly, poorly designed and implemented REDD+ actions may fail to deliver biodiversity benefits and in some cases may also cause negative impacts. 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 on biodiversity. Trade-offs between carbon and biodiversity outcomes can occur both locally and at wider spatial scales and are an important consideration to be addressed in REDD+ planning and implementation.

It is not easy to anticipate or measure all impacts of management actions on carbon and biodiversity, particularly as impacts can occur outside the area of management or in the future, and they can also evolve over time. Impacts of REDD+ interventions are also likely to vary significantly across different forest types and landscape conditions. Therefore, caution is needed when extrapolating management recommendations across different ecosystems, and the development of regionally-tailored strategies for REDD+ remains a major priority for future research.

Data on spatial patterns of biodiversity, expected trends in forest cover and condition, and on the effectiveness of existing management actions are needed to provide a better understanding of the full range of impacts of different REDD+ actions and guide decisions. 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. While the primary objective of REDD+ remains climate change mitigation, assessments of the impacts on biodiversity, ecosystem services and benefits to people as well as the governance factors that may affect implementation should be integrated into REDD+ decision-making.

6.4 Socio-economic and environmental trade-offs and synergies

The way in which REDD+ is implemented will determine its social and economic impacts on people, and a consideration of these impacts should be included early on in REDD+ implementation. REDD+ may generate substantial positive impacts but it may also lead to changes in resource management and access that will

disproportionately affect the poor and those that are most vulnerable.

Evidence suggests that pursuing social objectives alongside REDD+ will not only make the process more equitable but will also increase the likelihood of achieving carbon and biodiversity goals. For instance, increasing agricultural productivity can in some cases lead to reduced deforestation, and be a very powerful poverty-reduction tool. A real commitment to social and economic outcomes within REDD+ is essential, including considerations of governance and institutional arrangements, local engagement and participation, finance and markets, and timeframe.

Evidence shows that security of tenure, and associated authority for local decision-making, support better environmental management and the realisation of livelihood benefits. Inadequate recognition of such rights excludes the rural poor from decision making, and denies them access to potential benefits from market-based interventions, such as payments for ecosystem services. Weak tenure security also facilitates ‘land grabbing’ and other irregularities related to land ownership and transfer, which typically result in expropriation of lands from the most vulnerable groups. For REDD+ implementation to be effective and sustainable, tenure and property rights, including rights of access, use and ownership, need to be clear.

In many instances, true ‘win-win-win’ outcomes that are beneficial to biodiversity, carbon and people are not always available and there are sometimes difficult trade-offs. In these situations, a careful and inclusive evaluation should explicitly consider the following possible courses of action: (a) acknowledge the negative social/economic consequences, but do nothing about them, with subsequent repercussions for stakeholders; (b) compensate the losers, but acknowledge and accept social losses and disruption; (c) compensate the losers, and invest in secure, alternative livelihoods to attempt to offset some social losses; or (d) abandon the projects because of identified high human costs. Participatory and inclusive decision making may help identify the most appropriate choices in particular contexts, and avoid adverse consequences for the most vulnerable groups, including indigenous communities and women.

Socio-economic safeguards will help to avoid the most negative social and economic consequences, but for safeguards to be effective, social impacts need to be monitored carefully, especially access, authority and distributional issues. If REDD+ hopes to address the social and economic mechanisms that produce inequitable outcomes for vulnerable populations, it would need to go beyond the protective approach of safeguards and give greater parity to social objectives.

The landscape level provides a good scale to address and reconcile environmental, social and economic considerations relevant to REDD+. An integrated landscape management approach provides a framework to assess land use scenarios with REDD+ actions, their likely impacts on stakeholders and helps to define resulting trade-offs. Careful and inclusive spatial planning can positively

influence the distribution of winners and losers across the landscape so that REDD+ acts in the interests of the most vulnerable groups.

6.5 REDD+ and governance

REDD+ emerged within the UNFCCC but intersects with a wide array of other institutions involving different actors and priorities. Within this broader governance landscape, a diverse institutional complex has coalesced around REDD+ to govern an equally diverse array of REDD+ activities.

At the international level, three key sources of authority can be identified within this REDD+ complex: (inter-)governmental, fund-based and market-based. Each source offers different opportunities and constraints. (Inter-)governmental bodies have achieved few significant binding commitments regarding sustainable management of forests and biodiversity conservation. Fund-based institutions have been constrained by the quantity of funds available. At the same time REDD+ countries have limited capacity to meet the diverse operational requirements of REDD+ funders and absorb whatever finance is available. Market-based approaches are limited by the small size of the voluntary market.

REDD+ governance has also been shaped by autonomous national and local-level action. National governments play a critical role in establishing legal frameworks and tenure arrangements that can enable or constrain international efforts such as REDD+ as well as strengthen or undermine local authority. Without sufficient emphasis on local participation, there is a risk that REDD+ could recentralise government decision-making through policies and measures- for example, national targets to expand protected areas or increased regulatory enforcement- that undermine community-based forest governance.

The source of authority and scale of decision-making interact to shape the relative influence of intergovernmental processes, public and private donors, markets, national and sub-national governments and local populations on land use and management of forests. They also affect the means by which monitoring and reporting are incorporated into REDD+ projects. All of these factors in turn shape how the procedural and distributive benefits of REDD+ are shared among global to local actors.

Given the trade-offs involved in balancing power among different actors and institutions at different scales, the governance of forest management and biodiversity within and outside of REDD+ will continue to be pluralistic, involving multiple and competing forms of international to local rule-making. As observed in the previous GFEP report on forest governance², the most effective way forward may be to better understand, embrace and engage with this complexity rather than attempt to impose singular solutions.

In some cases, governance and policies independent of the REDD+ mechanism may be as, or more, important

than REDD+ in achieving carbon, forest management, biodiversity and social objectives. Hence REDD+ interventions should take care not to undermine initiatives and governance arrangements that are already working. Instead, REDD+ should aim to balance conflicting demands for international standardisation, national sovereignty, decentralisation and the empowerment of local communities.

6.6 Knowledge gaps

In the process of synthesising and analysing existing scientific knowledge on the various aspects of REDD+, a number of important knowledge gaps emerged which should be addressed as a matter of priority for effective implementation of REDD+ and related forest management interventions. These knowledge gaps are highlighted below.

Significant gaps exist in our understanding of the relationships between biodiversity and ecosystem functioning and provision of forest ecosystem services, including carbon sequestration, and how these relationships are affected by forest condition. Further work is needed to better understand:

- Relationships between plant species richness, functional diversity and biomass accumulation in diverse tropical forest systems;
- Relationships between species richness and ecosystem resistance (to disturbance);
- How the loss of forest biodiversity affects ecosystem processes;
- Long-term effects of forest ecosystem degradation on rates of recovery of forest ecosystems;
- Degradation/disturbance thresholds or tipping points beyond which recovery of ecosystem functions and provision of services may be severely constrained;
- The magnitude and dynamics of below-ground carbon stocks and fluxes in different forest types, as well as the time scales and the factors influencing the rates of recovery of biodiversity and carbon in disturbed, degraded, and secondary forests;
- The levels of ecosystem service provision from secondary forests, including increasingly widespread 'novel' forest ecosystems.

As regards management interventions under REDD+, existing knowledge is incomplete, particularly with respect to the:

- Differences in biodiversity impacts of REDD+ actions in different forest types;
- Impacts of management actions in relation to forest product extraction; impacts on tropical dry and swamp forests seem to be particularly poorly studied;
- Indirect effects of management interventions on forest and non-forest ecosystems at landscape and larger scales;

² Rayner, J., Buck, A. and Katila, P., 2011. *Embracing complexity: Meeting the challenges of international forest governance*. Vienna: International Union of Forest Research Organizations.

- Scaling up of existing knowledge and spatial data to guide management recommendations across different forest types, and develop regionally-tailored strategies for REDD+;
- Design of monitoring and assessment protocols that can provide cost-effective data on the performance of REDD+ initiatives for conserving both carbon and biodiversity.

Significant knowledge gaps remain in a number of areas that are important for understanding the social and economic consequences of REDD+ and biodiversity strategies. This is in part because of the complexity of the interactions between people, biodiversity and ecosystem services. Further research would be particularly needed related to:

- Decision-making methods and tools that help to assess social and economic impacts and consider and incorporate the interests of diverse stakeholders involved in or impacted by ecosystem-based interventions;
- Impacts of deforestation and degradation on the livelihoods of the poor;
- The relationship between agricultural intensification and employment, and the specific policy approaches that could mitigate or reverse job losses in this context;
- Systematic studies that draw lessons from project level REDD+ pilots to build evidence about governance and institutional issues;
- Evidence of how behaviour might change when PES-like strategies are implemented at a large scale on state or communal land;
- Assessment of how incentives within a payment-based REDD+ might function in conjunction with, for instance, either state or community-led protected area management;
- Greater understanding of what local stakeholders consider to be fair and beneficial outcomes from forest and landscape interventions.

In order to generate widespread policy learning and buy in, research on REDD+ should draw on a mix of methodologies, from systematic, large-scale comparative studies, to in-depth ethnographic field-work, to community-driven monitoring and evaluation. As concerns governance arrangements, research is needed in the following areas:

Assessments of the equity (e.g. balance of decision-making power), the economic efficiency and the on-the-ground effectiveness of different forms of REDD+ governance, across institutions and scales, and as relevant to multiple objectives. This includes research addressing the following sub-questions:

- How do different institutions and sources of authority interact with each other, and with what consequences?
- How can different sources of authority be combined to achieve results, and what are the trade-offs involved?
- Does the sharing of benefits from REDD+ translate into more effective biodiversity conservation, forest management and poverty alleviation?

Furthermore, given limited information concerning the funding of research related to REDD+, there is a clear

need to design and implement a survey targeting national and international research organisations, donors, the scientific community and other relevant stakeholders that could be used to collect, synthesise and evaluate data on REDD+ research funding in a systematic and comprehensive manner - a task that the Collaborative Partnership on Forests (CPF) might consider facilitating.

6.7 Moving forward with REDD+

Reducing the rates of global forest loss and degradation may yield unprecedented gains for both climate change mitigation and biodiversity conservation. It could also achieve significant social and economic gains. The degree to which these goals are achieved through a mechanism such as REDD+ will depend on whether and how REDD+ is translated into specific policies and practices that also contribute to biodiversity conservation and people's well-being. Should these two additional dimensions not be suitably addressed, there is a substantial risk that REDD+ may fail to deliver on all fronts.



Upland rice cultivation in the Western Ghats of India (Chikmagalur District, Karnataka).
Photo © John A. Parrotta



Appendix I

Overview of research funding for the different elements of the assessment

Lead author: Markku Kanninen

Currently, there is no information or data available that would allow us to make an accurate estimation of overall research investment in REDD+ and related fields. Presenting a comprehensive overview and an accurate estimate of research funding for the different elements of the present assessment is a challenging task. Firstly, REDD+ research covers numerous disciplines and topics, which makes it difficult to distinguish between 'REDD+ research' and 'research relevant to REDD+'. Secondly, several REDD+ initiatives (e.g. REDD+ demonstration activities) may have research components or elements of research in their work, but these are not explicitly classified as 'research'. Finally, the question can be posed "should basic research on methods and technology development (e.g. for forest assessment and monitoring, including satellite technology etc.) be classified as 'REDD+ research' or not?"

Forest biodiversity, carbon and other ecosystem services: relationships and impacts of deforestation and forest degradation

Research related to the first element of this study 'The relationship between forest biodiversity, carbon and other ecosystem services' includes the development of methods and tools for the characterisation of different types of forests, and for the assessment of their biodiversity, and carbon pools and fluxes. In addition, it includes the actual assessment and monitoring (including remote sensing and field studies) of land use and land use change, biodiversity, forest resources, and changes in carbon pools and fluxes. Thus, it is difficult to draw the line between 'general' and 'REDD+ specific' research. However, the emergence of REDD+ and interest in the role of forests in climate change mitigation during the past decade has boosted investments in research on land use and land use change, biodiversity, forest carbon assessment, and

on forest inventory and monitoring (e.g. IPCC, 2003; IPCC, 2006; Gardner, 2010; GOF-C-GOLD, 2010). International organisations and specialised agencies (e.g. FAO, the European Commission's Joint Research Centre, and numerous national agencies worldwide) have made considerable investments in the development of remote sensing (e.g. satellite technology) and inventory methods for monitoring land use changes and forests that are an essential part of monitoring, reporting and verification (MRV) of REDD+.

Impacts of forest and land management on biodiversity and carbon

While research on 'Impacts of forest and land management on biodiversity and carbon' is not recent, a renewed interest in this topic can be traced back to the post-Rio Summit (1992) (e.g. Putz and Pinard, 1993; Pinard and Putz, 1997; Putz et al., 2001). After the emergence of REDD and REDD+ some five years ago, research on the relationships between forest management, biodiversity and forest carbon has become more systematic and intensive (e.g. GOF-C-GOLD, 2010; Gardner et al. 2012).

Social and economic considerations relevant to REDD+

Research on 'Social and economic considerations relevant to REDD+' is cross-sectoral and thus, includes work on a vast array of topics straddling several disciplines, from the role of forest and biodiversity for livelihoods, to issues related to payment and reward schemes for ecosystem services. Research on some of these issues dates back to the Eighth World Forestry Congress in 1978, and has since produced relevant results on forests and communities and on deforestation (e.g. Ostrom, 1990; Kaimowitz and Angelsen, 1998). On the other hand, the research on ecosystem services is a new and rapidly expanding field (Nicholson et al., 2009) with a rapidly increasing funding

base, which can contribute significantly to the formulation of REDD+ policies and implementation mechanisms in the future.

Governance for REDD+, forest management and biodiversity: Existing approaches and future options

Research related to ‘Governance for REDD+’ emerged with the inclusion of REDD into the UNFCCC negotiation process at COP 13 (e.g. Kanninen et al., 2007; Eliasch, 2008; Angelsen, 2008; Angelsen et al., 2009). Funding for this research has increased rapidly during the last years due to strong donor support, e.g. Norway and other REDD Alliance members, and from the private sector and foundations (e.g. CLUA – Climate and Land Use Alliance). However, there are several areas of research that need to be strengthened to support REDD+ policy formulation, e.g. the role of access rights and tenure and of local institutions, inclusion of women, indigenous people and the importance of forests to local livelihoods.

Due to the fact that the research on REDD+ is versatile and its agenda is constantly expanding, there is not only a need to increase research investments in these fields, but simultaneously, a need to build human capacity for this new and expanding research. This is particularly the case of the developing world – the REDD+ countries themselves. Paradoxically, in spite of all the increased investments in REDD+ research worldwide, the capacity gap for REDD+ research between developed and developing countries is getting larger rather than smaller.

In conclusion, there is a clear need to design and implement a survey targeting national and international research organisations, donors, the scientific community and other relevant stakeholders that could be used to collect, synthesise and evaluate data on REDD+ research funding in a systematic and comprehensive manner - a task that CPF might consider facilitating.

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Appendix 2

Glossary of terms and definitions used in the assessment report

Adaptation (climate): Adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities (Seppälä et al., 2009)

Adaptive capacity (in relation to climate change impacts): The ability of a system to adjust to climate change (including climate variability and extremes) to moderate potential damages, to take advantage of opportunities, or to cope with the consequences (IPCC, 2007).

Adaptive management: A dynamic approach to forest management in which the effects of treatments and decisions are continually monitored and used, along with research results, to modify management on a continuing basis to ensure that objectives are being met (IUFRO, 2005).

Above-ground biomass (AGB): All biomass of living vegetation, both woody and herbaceous, above the soil including stems, stumps, branches, bark, seeds, and foliage (FAO, 2004; IPCC, 2006).

Above-ground biomass growth: Oven-dry weight of net annual increment of a tree, stand or forest plus oven-dry weight of annual growth of branches, twigs, foliage, top and stump. The term “growth” is used here instead of “increment”, since the latter term tends to be understood in terms of merchantable volume (IPCC, 2006).

Afforestation: Establishment of forest through planting and/or deliberate seeding on land that, until then, was not classified as forest (FAO, 2010). According to the definition used by the UNFCCC, afforestation can take place on land that has not been covered by forest for at least 50 years. See also *Reforestation*.

Agroforestry: A collective name for land use systems and practices in which woody perennials are deliberately integrated with crops and/or animals on the same land management unit. The integration can be either in a spatial mixture or in a temporal sequence. There are normally both ecological and economic interactions between woody and non-woody components in agroforestry (World Agroforestry Centre).

Alien invasive species: see *Invasive alien species*

Assisted natural regeneration: The natural regeneration with human assistance through removal of external pressures, such as weeds and biotic interference and sometimes application of controlled disturbances to trigger germination of native species such as mosaic and or ecological burns or preparation of the germination site, enabling the inherent resilience of the site to naturally regenerate the native species. (FAO, 2004).

Below-ground biomass (BGB) : All biomass of live roots. Fine roots of less than (suggested) 2mm diameter are often excluded because these often cannot be distinguished empirically from soil organic matter or litter. (FAO, 2004; IPCC, 2006).

Biodiversity (Biological diversity): The variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems. (CBD, Article 2).

Biomass: Live organic material both above-ground and below-ground, e.g. trees, crops, grasses, roots. Biomass includes the pool definition for above- and below-ground biomass (adapted from: IPCC, 2003; FAO, 2004). See also *Above-ground biomass*, *Below-ground biomass*.

Biome: Major and distinct regional element of the biosphere, typically consisting of several ecosystems (e.g., forests, rivers, ponds, swamps) within a region of similar climate. Biomes are characterised by typical communities of plants and animals (IPCC, 2007).

Carbon: In this assessment report, except when referring to specific [carbon] stocks and fluxes, ‘carbon’ refers to the net balance of CO₂ and non-CO₂ greenhouse gas emissions and removals.

Carbon cycle: The term used to describe the flow of carbon (in various forms, e.g. as carbon dioxide) through the atmosphere, ocean, terrestrial biosphere and lithosphere (IPCC, 2007).

Carbon balance: see *Net ecosystem carbon balance*.

Carbon sequestration: The process of increasing the carbon content of a reservoir/pool other than the atmosphere (IPCC, 2007).

Carbon sink: Any process, activity, or mechanism that removes a greenhouse gas, an aerosol, or a precursor of a greenhouse gas or aerosol from the atmosphere (IPCC, 2007).

Carbon source: Any process, activity, or mechanism that releases a greenhouse gas, an aerosol, or a precursor of a greenhouse gas or aerosol into the atmosphere (IPCC, 2007).

Carbon emission: see *Emission*

Carbon stock/reservoir: A component of the climate system, other than the atmosphere, that has the capacity to store, accumulate or release a substance of concern (e.g., carbon or a greenhouse gas). Oceans, soils and forests are examples of carbon reservoirs (IPCC, 2007). More simply, the quantity of carbon in a pool.

Carbon stock change: The carbon stock in a pool changes due to gains and losses. When losses exceed gains, the stock decreases, and the pool acts as a source; when gains exceed losses, the pools accumulate carbon, and the pools act as a sink.

Carbon storage: see *Carbon sequestration*

Climate change: Refers to any change in climate over time, whether due to natural variability or as a result of human activity. This usage differs from that in the United Nations Framework Convention on Climate Change (UNFCCC), which defines *climate change* as: ‘a change of climate which is attributed directly or indirectly to human activity that alters the composition of the global atmosphere and which is in addition to natural climate variability observed over comparable time periods’ (IPCC, 2007).

Dead wood: Includes all non-living woody biomass not contained in the litter, either standing, lying on the ground, or in the soil. Dead wood includes wood lying on the surface, dead roots, and stumps larger than or equal to 10 cm in diameter or any other diameter used by the country (IPCC, 2003).

Deforestation: The conversion of forest to another land use or the long-term reduction of the tree canopy cover below the minimum 10% threshold (FAO, 2010). Deforestation implies the long-term or permanent loss of forest cover and implies transformation into another land use. Such a loss can only be caused and maintained by a continued human-induced or natural perturbation. Deforestation includes areas of forest converted to agriculture, pasture, water reservoirs and urban areas. The term specifically *excludes* areas where the trees have been removed as a result of harvesting or logging, and where the forest is expected to regenerate naturally or with the aid of silvicultural measures. Deforestation also includes areas where, for example, the impact of disturbance, overutilisation or changing environmental conditions affects the forest to an extent that it cannot sustain a tree cover above the 10% threshold (FAO, 2001).

Degradation: see *Forest degradation*

Displacement factor: The amount of greenhouse gas emission reduction per unit of biomass carbon use (Sathre and O’Connor, 2010), through, e.g., (1) the conversion of harvested biomass to end products, minimising waste, (2) end

products used to substitute other emissions-intensive materials, e.g. steel or concrete in building construction, and (3) end products used in a cascading system that emphasises reuse, recycling and responsible use of wood products.

Ecological restoration: Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SERI, 2004).

Ecological resilience: The ability of a system to absorb impacts before a threshold is reached where the system changes into a different state. (Gunderson, 2000)

Ecological threshold: An ecological threshold is the point at which there is an abrupt change in an ecosystem quality, property or phenomenon, or where small changes in an environmental driver produce large and unpredictable responses in the ecosystem (Groffman et al., 2006).

Ecosystem: A dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit (CBD, 1992).

Ecosystem state: The recognisable condition of an ecosystem under a given set of biotic and abiotic conditions that includes the typical suite of species and that is stable in ecological time. Multiple stable states are possible for a given set of conditions (Holling, 1973).

Ecosystem (or ecological) functions: Ecosystem ‘functions’ are synonymous with ‘processes’ and refer to all of the physical, chemical and biological actions performed by organisms within ecosystems. Some of these functions are ecosystem services, including production, pollination, nutrient cycling (e.g., decomposition, N₂-fixation) and carbon storage (MA, 2005) that directly benefit humans. Other examples include photosynthesis, predation, scavenging and herbivory.

Ecosystem processes: see *Ecosystem functions*

Ecosystem resistance: The capacity of an ecosystem to absorb disturbances and remain largely unchanged (Holling, 1973).

Ecosystem services: Ecological processes or functions having monetary or non-monetary value to individuals or society at large. There are (i) supporting services such as productivity or biodiversity maintenance, (ii) provisioning services such as food, fibre or fish, (iii) regulating services such as climate regulation or carbon sequestration, and (iv) cultural services such as tourism or spiritual and aesthetic appreciation (IPCC, 2007).

Ecosystem stability: The capacity of an ecosystem to remain more or less in the same state within bounds, that is, the capacity to maintain a dynamic equilibrium in time while resisting change (Holling, 1973).

Edge effect: The tendency for increased variety and density at community junctions (Odum, 1953). The effect of processes, both abiotic and biotic, at the edge of ecosystems that result in a detectable difference in composition, structure, or function, as compared with the ecosystem on either side of the edge (Harper, et al., 2005).

Emission: The release of greenhouse gases and/or their precursors into the atmosphere over a specified area and period of time (IPCC, 2003).

Endemic species: A native species restricted to a particular geographic region owing to factors such as isolation or in response to soil or climatic conditions. (CBD: <http://www.cbd.int/forest/definitions.shtml>).

Enrichment planting: The improvement of the percentage of desirable species or genotypes or increasing biodiversity in a forest by interplanting (Helms, 1998).

Forest: Land spanning more than 0.5 hectares with trees higher than 5 metres and a canopy cover of more than 10 percent, or trees able to reach these thresholds *in situ*. It does not include land that is predominantly under agricultural or urban land use (FAO, 2010). Forests include both *natural forests* (sensu CPF, 2005) and *planted forests* (sensu FAO, see below). It also includes areas temporarily unstocked, e.g. after disturbance, that are expected to revert back to forest.

Forest ecosystem: A forest ecosystem can be defined at a range of scales. It is a dynamic complex of plant, animal and micro-organism communities and their abiotic environment interacting as a functional unit, where trees are a key component of the system. Humans, with their cultural, economic and environmental needs are an integral part of many forest ecosystems. (CBD: <http://www.cbd.int/forest/definitions.shtml>).

Forest degradation: The reduction of the capacity of a forest to provide goods and services.

Note: A degraded forest delivers a reduced supply of goods and services from a given site and maintains only limited biological diversity. It has lost the structure, function, species composition and/or productivity normally associated with the natural forest type expected at that site (ITTO, 2002).

Forest dependent people: Encompasses peoples and communities that have a direct relationship with forests and trees and live within or immediately adjacent to forested areas, and depend on them for their subsistence (FAO, 1996).

Forest fragmentation: Any process that results in the conversion of formerly continuous forest into patches of forest separated by non-forested lands (CBD: <http://www.cbd.int/forest/definitions.shtml>).

Forest landscape restoration: A planned process that aims to regain ecological integrity and enhance human well-being in deforested or degraded landscapes (WWF and IUCN, 2000; Mansourian et al., 2005).

Forest management: The processes of planning and implementing practices for the stewardship and use of forests and other wooded land aimed at achieving specific environmental, economic, social and/or cultural objectives. Includes management at all scales such as normative, strategic, tactical and operational level management (FAO, 2004).

Forest rehabilitation: The process of restoring the capacity of a forest to provide goods and services again, where the state of the rehabilitated forest is not identical to its state before (CPF, 2005).

Forest restoration: Management applied in degraded forest areas which aims to assist the natural processes of forest recovery in a way that the species composition, stand structure, biodiversity, functions and processes of the restored forest will match, as closely as feasible, those of the original forest (IUFRO, 2005). See also *Assisted natural regeneration*, *Ecological restoration* and *Forest landscape restoration*.

Functional groups: Assemblages of species performing similar functional roles within an ecosystem, such as pollination, production, or decomposition (i.e., trophic groups), hence providing some redundancy. (Hooper and Vitousek, 1997)

Governance: Any effort to coordinate human action towards goals. In the common distinction between government and governance, the latter is usually taken to refer specifically to coordination mechanisms that do not rest on the authority and sanctions possessed by states (Stoker, 1998), but the report uses 'governance' in the broadest sense of coordination.

Greenhouse gas: Gaseous constituents of the atmosphere, both natural and anthropogenic, that absorb and emit radiation at specific wavelengths within the spectrum of infrared radiation emitted by the Earth's surface, the atmosphere, and clouds. This property causes the greenhouse effect. Water vapour (H₂O), carbon dioxide (CO₂), nitrous oxide (N₂O), methane (CH₄) and ozone (O₃) are the primary greenhouse gases in the Earth's atmosphere. As well as CO₂, N₂O and CH₄, the Kyoto Protocol deals with the greenhouse gases sulphur hexafluoride (SF₆), hydrofluorocarbons (HFCs) and perfluorocarbons (PFCs) (IPCC, 2007).

Gross primary production (GPP): Ecosystem-level photosynthetic gain of CO₂-C (Chapin et al., 2006).

Habitat: The geographical unit that effectively supports the survival and reproduction of a given species or of individuals of a given species; the composite of other organisms as well as abiotic factors therein describe the geographical unit.

Habitat loss: Used with reference to an individual species, is the permanent conversion of former (forest) habitat to an area where that species can no longer exist, be it still forested or not (CBD: <http://www.cbd.int/forest/definitions.shtml>).

Indigenous species: see *Native species*

Introduced species: A species, subspecies or lower taxon, occurring outside its natural range (past or present) and dispersal potential (i.e. outside the range it occupies naturally or could occupy without direct or indirect introduction or care by humans) (FAO, 2004). *Note:* synonymous with 'exotic' or 'alien' species.

Invasive alien species: Any species that are non-native to a particular ecosystem and whose introduction and spread causes, or are likely to cause, socio-cultural, economic or environmental harm or harm to human health (FAO, 2008).

Invasive species: Organisms (usually transported by humans) which successfully establish themselves in, and then overcome pre-existing native ecosystems (IUFRO, 2005).

Land sparing: The promotion of agricultural techniques that encourage the highest possible yields in a given area (even if it involves reduced in-farm biodiversity) with the goal of meeting agricultural needs in the minimum possible area, so as to reduce the pressure over wild areas.

Land sharing: The promotion of agricultural techniques, mainly agroforestry, that are ‘friendly’ to wild species, aimed at fostering the co-existence of managed (crops or livestock) and wild species in the same area.

Leakage: In the REDD+ context, ‘leakage’ refers to direct emissions elsewhere caused by the emission reduction in a project/programme area, e.g., protection of a forest area in one location leading to emissions caused by deforestation in other locations.

Mitigation (climate): An anthropogenic intervention to reduce the anthropogenic forcing of the climate system; it includes strategies to reduce greenhouse gas sources and emissions and enhancing greenhouse gas sinks (IPCC, 2007).

Monoculture: see *Monotypic stand*

Monotypic stand: A forest stand containing one tree species (Thompson et al., 2009).

Native species: Species which naturally exists at a given location or in a particular ecosystem, i.e. it has not been moved there by humans. (CBD: <http://www.cbd.int/forest/definitions.shtml>).

Natural forest: Forest stands composed predominantly of native tree species established naturally [i.e., through natural regeneration]. This can include assisted natural regeneration, excluding stands that are visibly offspring/descendants of planted trees (CPF, 2005). See also *Primary forest*, *Naturally regenerated forest*, *Secondary forest*.

Naturally regenerated forest: Forest predominantly composed of trees established through natural regeneration (FAO 2010). See also *Primary forest*, *Secondary forest*.

Net biome productivity (NBP): The net ecosystem carbon balance or net change in ecosystem carbon stocks due to all causes over a large region (Chapin et al., 2006).

Net ecosystem carbon balance (NECB): the overall ecosystem C balance from all sources and sinks - physical, biological, and anthropogenic for a specified area over a specified time (Chapin et al., 2006). NECB is reported from the ecosystem perspective, thus a forest sink has a positive sign (an increase in ecosystem C stocks) and a forest source a negative sign (a reduction in ecosystem C stocks).

Net ecosystem exchange (NEE): The net CO₂ flux from the ecosystem to the atmosphere, including fire emissions. NEE is reported from the perspective of the atmosphere, thus a forest sink has a negative sign (a loss from the atmosphere) and a forest source positive sign (a gain for the atmosphere) (Chapin et al., 2006).

Net ecosystem production (NEP): (a) the difference between ecosystem-level photosynthetic gain of CO₂-C (gross primary production, or GPP) and ecosystem (plant, animal, and microbial) respiratory loss of CO₂-C (ecosystem respiration, or ER), or (b) the net rate of C accumulation in ecosystems prior to the impacts of disturbances (Woodwell and Whittaker, 1968). NEP is reported from the ecosystem perspective, thus a forest sink has a positive sign (an increase in ecosystem C stocks) and a forest source a negative sign (a reduction in ecosystem C stocks).

Net primary production (NPP): Net primary production is the rate of photosynthesis minus the rate of respiration of primary producers (autotrophic respiration).

Non-timber forest products (NTFP): All biological materials other than timber, which are extracted from forests for human use. Forest refers to a natural ecosystem in which trees are a significant component. In addition to trees forest products are derived from all plants, fungi and animals (including fish) for which the forest ecosystem provides habitat (IUFRO, 2005).

Novel ecosystems: Ecosystems that differ in composition and/or function from present and past systems as a consequence of changing species distributions and environmental alteration through climate and land use change. (Hobbs et al., 2009).

Payments for Ecosystem (or Environmental) Services (PES): A type of economic incentive offered for those that manage ecosystems (including agricultural lands) to improve the flow of environmental services that they provide. These incentives can be provided by all those who benefit from environmental services, including local, regional and global stakeholders. REDD+ can be understood as a global PES scheme.

Planted forest: Forest predominantly composed of trees established through planting and/or deliberate seeding (FAO, 2010). Includes forests resulting from afforestation, reforestation, and some forms of forest restoration (see also for *Afforestation, Ecological restoration, Forest restoration, Forest landscape restoration, Reforestation*).

Planted forest of introduced species: Planted forests in which the planted/seeded trees are predominantly of introduced species, i.e., species, subspecies or lower taxon, occurring outside its natural range (past or present) and dispersal potential (i.e. outside the range it occupies naturally or could occupy without direct or indirect introduction or care by humans) (FAO, 2010). *Note:* introduced species is synonymous with ‘exotic species’ and ‘alien species’.

Plantation forests: Planted forests that have been established and are (intensively) managed for commercial production of wood and non-wood forest products, or to provide a specific environmental service (e.g. erosion control, landslide stabilisation, windbreaks, etc.) (Carle and Holmgren, 2003).

Policy instruments (= policy tools): Tools designed to regulate citizens’ behaviour and define their legal rights. Substantive policy instruments direct government intervention that required or motivated a certain course of behavioural change. They comprise regulatory (e.g., prescriptions, proscriptions), financial (e.g., subsidy, taxation) and informational (e.g., education) policy means, which act directly on the addressees. Procedural policy instruments act on the process indirectly through institutional or organisational means by which policy is created (IUFRO, 2005).

Primary forest: Naturally regenerated forest of native species, where there are no clearly visible indications of human activities [including commercial logging] and the ecological processes are not significantly disturbed (FAO, 2010).

Production: see *Gross primary production, Net primary production* and *Net ecosystem production*.

Rebound effect: The phenomenon whereby increased productivity of an economic activity leads to a net increase in the use of a certain input (in this case land). This happens when the activity becomes so much more attractive that the consequent increase in production outweighs the gains in productivity, leading to a net increase in the demand for that input. In this case this would mean increased deforestation.

Reduced impact logging: The intensively planned and carefully controlled implementation of timber harvesting operations to minimise the environmental impact on forest stands and soils (International Tropical Timber Organization).

Redundancy: The concept of ecological redundancy is sometimes referred to as *functional compensation* and assumes that more than one species performs a given role within an ecosystem (Walker, 1992) More specifically, it is characterised by a particular species increasing its efficiency at providing a service when conditions are stressed in order to maintain aggregate stability in the ecosystem (Frost et al., 1995)

Reforestation: Re-establishment of forest through planting and/or deliberate seeding on land classified as forest after a temporary period (< 10 years) during which there was less than 10 percent canopy cover due to human-induced or natural perturbations (adapted from FAO, 2010). According to the definition used by the UNFCCC, reforestation can occur on land that was forested but that has been converted to non-forested land.

Resilience: The ability of a social or ecological system to absorb disturbances while retaining the same basic structure and ways of functioning, the capacity for self-organisation, and the capacity to adapt to stress and change (IPCC, 2007). See also *Ecological resilience*.

Resistance (see *Ecosystem resistance*)

Secondary forest: forests regenerating largely through natural processes after significant removal or disturbance of the original forest vegetation by human or natural causes at a single point in time or over an extended period, and displaying a major difference in forest structure and/or canopy species composition with respect to pristine primary forests (FAO, 2003). Categories of secondary forest include:

Post extraction secondary forests: forests regenerating largely through natural processes after significant reduction in the original forest canopy through tree extraction at a single point in time or over an extended period and displaying

a major change in forest structure and/or canopy species composition from that of the primary/natural forests on similar site conditions in the area given a long time without significant disturbance.

Swidden fallow secondary forests: forests regenerating largely through natural processes in woody fallows of swidden agriculture for the purpose of food production by farmers and/or communities.

Rehabilitated secondary forests: forests regenerating largely through natural processes on degraded lands. Regeneration could be enhanced by protection from chronic disturbance, site stabilisation, water management and enrichment planting to facilitate natural regeneration.

Post-fire secondary forests: forests regenerating largely through natural processes after significant reduction in the original forest canopy caused by fires at a single point in time or over an extended period, and displaying a major change in forest structure and/or canopy species composition relative to those of potential primary/natural forests on similar site conditions in the area, given a long time without significant disturbance.

Post-abandonment secondary forests: forests regenerating largely through a natural process after abandonment of alternative land uses such as agriculture or pasture development for cattle production (FAO, 2003).

Sink: see *Carbon sink*

Slash-and-burn cultivation: see *Shifting cultivation*

Source: see *Carbon source*

Swidden agriculture: see *Shifting cultivation*

Shifting cultivation: Also referred to as *slash-and-burn cultivation* or *swidden agriculture*, is difficult to define precisely, since it is perceived and used by different people in different contexts in widely differing ways. The essential characteristics of shifting cultivation are that an area of forest is cleared, usually rather incompletely, the debris is burnt, and the land is cultivated for a few years - usually less than five - then allowed to revert to forest or other secondary vegetation before being cleared and used again (FAO, 1984). In areas of shifting agriculture, forest, forest fallow and agricultural lands appear in a dynamic pattern where deforestation and the return of forest occur frequently in small patches.

Soil carbon: Organic carbon in mineral and organic soils (including peat) to a specified depth chosen by the country and applied consistently through the time series. Live fine roots of less than 2 mm (or other value chosen by the country as diameter limit for below-ground biomass) are included with soil organic matter where they cannot be distinguished from it empirically (IPCC, 2006).

Soil organic matter: Includes organic carbon in mineral soils to a specified depth chosen by the country and applied consistently through the time series. Live and dead fine roots and dead organic matter within the soil, that are less than the minimum diameter limit (suggested 2mm) for roots and dead organic matter, are included with soil organic matter where they cannot be distinguished from it empirically (IPCC, 2006).

Species diversity: A measure of the diversity within an ecological community that incorporates both species richness (the number of species in a community) and the evenness of species' abundances.

Species richness: The number of species present in a sample, community, or taxonomic group.

Stability: see *Ecosystem stability*.

Succession: Progressive changes in species composition and forest community structure caused by natural processes over time (Helms, 1998).

Sustainable forest management: A dynamic and evolving concept, aims to maintain and enhance the economic, social and environmental values of all types of forests, for the benefit of present and future generations. The seven thematic elements of sustainable forest management are: (a) extent of forest resources; (b) forest biological diversity; (c) forest health and vitality; (d) productive functions of forest resources; (e) protective functions of forest resources; (f) socio-economic functions of forests; and (g) legal, policy and institutional framework. The thematic elements are drawn from the criteria identified by existing criteria and indicators processes, as a reference framework for sustainable forest management (UN, 2007).

Threshold (see *Ecological threshold*)

Total biomass: Growing stock biomass of trees, stands or forests plus biomass of branches, twigs, foliage, seeds, stumps, and sometimes, non-commercial trees. Differentiated into above-ground biomass and below-ground biomass (IPCC, 2006).

Total biomass growth: Biomass of the net annual increment of trees, stands, or forests, plus the biomass of the growth of branches, twigs, foliage, seeds, stumps, and sometimes, non-commercial trees. Differentiated into above-ground biomass growth and below-ground biomass growth (IPCC, 2006).

Traditional (ecological) knowledge: A cumulative body of knowledge, practice and belief, handed down through generations by cultural transmission and evolving by adaptive processes, about the relationship between living beings (including humans) with one another and with their forest environment (Berkes, 1999; UNEP, 2008).

Trophic cascade: A dynamic ecosystem process, where removal of a predator results in a cascade of effects down a food chain, with inherent effects on ecosystem stability, often resulting in a change in state (Terborgh and Estes, 2010).

Vulnerability (ecosystem): The degree to which a system is susceptible to, and unable to cope with, adverse effects of climate change, including climate variability and extremes. Vulnerability is a function of the character, magnitude, and rate of climate change and variation to which a system is exposed, its sensitivity, and its adaptive capacity (IPCC, 2007).

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