Forest Resources Assessment Working Paper 177

Assessing forest degradation

Towards the development of globally applicable guidelines

November, 2011

Forest Resources Assessment Working Paper 177

Assessing forest degradation

Towards the development of globally applicable guidelines

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
Rome, 2011

Contents

Boxes

Tables

Figures

Contributors

Many people contributed to the production of this document. In particular, chapters were authored as follows:

Chapter 1: Victoria Heymell and Ken MacDicken.

Chapter 2: Ken MacDicken (based on an earlier paper by Markku Simula).

Chapter 3: Dan Altrell, Anne Branthomme and Rebecca Tavani

Chapter 4: Ian Thompson, Robert Nasi, Kimiko Okabe, Valerie Kapos and James Gordon

Chapter 5: Manuel R. Guariguata, Cesar Sabogal and Diji Chandrasekharan

Chapter 6: Des McGarry

Foreword

Forests provide us with a wide range of goods and services. Today, just as centuries ago, we still need forests for their products such as timber, paper, medical plants, fruits etc. Presently more people understand the values of services that forests provide, including wildlife habitat, hydrological functions and carbon storage.

Likewise the degradation of forest resources is an important society concern that is perceived in many different ways. Forest degradation can be a serious environmental, social and economic problem with the potential to adversely affect millions of people who depend on forest goods and services. Given the contribution of forests to sustainable development and their role for human well-being, the state of the forests is important to all of us.

Good information on the extent of forest degradation is needed to elaborate policies and implement forest-management plans allowing the restoration of degraded forests and the rehabilitation of degraded forest lands.

Identifying and assessing the condition of forests is not easy – particularly since people have widely different views of what constitutes degradation. For some, any forest management activity may cause degradation. For others forest is only degraded when it can no longer deliver needed goods and services. There is no globally agreed definition of forest degradation which makes the discussion more complex. FAO, together with members of the Collaborative Partnership have taken a number of steps to tackle this problem. Results of this work are summarized in a series of working papers that can be found at http://www.fao.org/forestry/fra/2560/en/.

This document pulls together a range of views and approaches to the assessment of forest degradation. It should be regarded as precursor to the development of comprehensive, globally applicable guidelines for assessing forest degradation. There is much work yet to be done on this important topic – we trust the present paper contributes to the goal of reducing and mitigating the inevitable processes of forest degradation.

Eduardo Rojas-Briales
Assistant Director-General, Forestry Department, FAO

Acknowledgements

The preparation of "Assessing forest degradation – towards the development of globally applicable guidelines" represents a joint effort of the Collaborative Partnership on Forests (CPF) undertaken under the umbrella of the 2010 Global Forest Resources Assessment (FAO, 2010). FAO gratefully acknowledges the financial and in-kind contributions to this work made by CPF partners.

Thanks go to all who have contributed to the development of this work: to those who provided answers to the initial questionnaires, to the authors of the 20 case studies, and to the authors and experts who contributed to technical meetings.

A special thanks go to the authors of the chapters that describe key criteria for measuring forest degradation and to Victoria Heymell who saw the task through many arduous steps. Thanks also go to Mette Løyche Wilkie, Emma Foti, Marisalee Palermo and the rest of the FRA team in FAO for providing advice, guidance and assistance along the way and to Alastair Sarre who did the final editing.

Acronyms

AAC	annual allowable cut
ASTER	Advanced Space-borne Thermal Emission and Reflection Radiometer
C&I	criteria and indicators
CBD	Convention on Biological Diversity
CPF	Collaborative Partnership on Forests
dbh	diameter at breast height
FAO	Food and Agriculture Organization of the United Nations
FMU	forest management unit
GIS	geographic information system
GPFLR	Global Partnership on Forest Landscape Restoration
IPCC	Intergovernmental Panel on Climate Change
ITTO	International Tropical Timber Organization
IUFRO	International Union of Forest Research Organizations
LiDAR	Light Detection and Ranging
MAI	mean annual increment
MERIS	Medium Resolution Imaging Spectroradiometer
MODIS	Moderate Resolution Imaging Spectroradiometer
NRV	natural range of variability
NWFP	non-wood forest product
REDD	reduction in emissions from deforestation and forest degradation
SFM	sustainable forest management
SPOT	Satellite Pour l'Observation de la Terre
UNEP-WCMC	United Nations Environment Programme-World Conservation Monitoring Centre
UNFCCC	United Nations Framework Convention on Climate Change

1. Introduction

WHAT IS FOREST DEGRADATION?

FAO (2002) defines forest degradation as:

> *The reduction of the capacity of a forest to provide goods and services*.

Perceptions of forest degradation are many and varied, depending on the driver of degradation and the goods or services of most interest. For example, a manager who replaces a natural forest with a plantation to supply desired wood products is unlikely to perceive his forest as degraded. On the other hand, his plantation is less capable of providing many of the goods and services that a fully functioning natural forest would provide on the same site, partly because of the reduced biodiversity generally associated with plantations, which to others would constitute a degraded state.

In a recent survey, Lund (2009) found more than 50 definitions of forest degradation, formulated for various purposes. FAO (2009) shows that many such definitions are either very general or their focus is on the reduction of productivity, biomass or biological diversity. Definitions that refer to multiple-use forests or multiple forest benefits may consider forest values comprehensively but are more difficult to apply universally in a consistent and transparent way.

From the perspective of international forest-related reporting, coherent, comparable and harmonized definitions are desirable. The development of such definitions is challenging, however, not least because national circumstances have implications for how international definitions can be applied. Nevertheless, the general definition of forest degradation given above provides an adequate umbrella at the international level and a common framework for developing more specific definitions for particular purposes. It is also compatible with an ecosystem-services approach.

Forest degradation involves a change process that negatively affects the characteristics of a forest such that the value and production of its goods and services decline. This change process is caused by disturbance (although not all disturbance causes degradation), which may vary in extent, severity, quality, origin and frequency. Disturbance may be natural (e.g. that caused by fire, storm or drought), human-induced (e.g. through harvesting, road construction, shifting cultivation, hunting or grazing) or a combination of the two. Human-induced disturbance may be intentional (direct), such as that caused by logging or grazing, or it may be unintentional (indirect), such as that caused by the spread of an invasive alien species (FAO, 2009).

Box 1.1 presents the main definitions of forest degradation by relevant international bodies. The generic definition of the Second Expert Meeting on Harmonizing Forest-related Definitions for Use by Various Stakeholders (FAO, 2002b), which is used in this document, provides a common framework for all the international definitions and is also compatible with the ecosystem-services approach. Chapter 2 presents some of the common and contrasting elements of various national-level definitions and discusses issues around what to assess in monitoring trends in forest degradation.

WHY DOES IT MATTER?

Forest degradation is a serious environmental, social and economic problem. Quantifying the scale of the problem is difficult, however, because forest degradation has many causes, occurs in different forms and with varying intensity, and is perceived differently by different stakeholders. The International Tropical Timber Organization (ITTO, 2002) estimated that up to 850 million hectares of tropical forest and forest lands could be degraded. The Global Partnership on Forest Landscape Restoration (GPFLR, undated) suggested that more than one billion hectares of deforested and degraded forest land worldwide are suitable and available for restoration.

BOX 1.1

International definitions of forest degradation/degraded forest

Organization	Definition
Second Expert Meeting on Harmonizing Forest-related Definitions for Use by Various Stakeholders (FAO, 2002b)	**Forest degradation** is the reduction of the capacity of a forest to provide goods and services.
FAO (2001) – Global Forest Resources Assessment 2000	**Forest degradation** is changes within the forest which negatively affect the structure or function of the stand or site, and thereby lower the capacity to supply products and/or services.
ITTO (2002, 2005)	**Forest degradation** refers to the reduction of the capacity of a forest to produce goods and services (ITTO, 2002). Capacity includes the maintenance of ecosystem structure and functions (ITTO, 2005). 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). Explanatory notes ((ITTO, 2002; 2005): Forests that have been altered beyond the normal effects of natural processes are categorized as either degraded primary forest, secondary forest, or degraded forest land. **Degraded primary forest:** primary forest in which the initial cover has been adversely affected by the unsustainable harvesting of wood and/or non-wood forest products so that its structure, processes, functions and dynamics are altered beyond the short-term resilience of the ecosystem; that is, the capacity of these forests to fully recover from exploitation in the near to medium term has been compromised. **Secondary forest:** woody vegetation regrowing on land that was largely cleared of its original forest cover (i.e. carried less than 10% of the original forest cover). Secondary forests commonly develop naturally on land abandoned after shifting cultivation, settled agriculture, pasture or failed tree plantations. **Degraded forest land:** former forest land severely damaged by the excessive harvesting of wood and/or non-wood forest products, poor management, repeated fire, grazing or other disturbances or land uses that damage soil and vegetation to a degree that inhibits or severely delays the re-establishment of forest after abandonment.

CBD (2001, 2005)	A **degraded forest** delivers a reduced supply of goods and services from the given site and maintains only limited biological diversity. Such a forest may have lost its structure, species composition or productivity normally associated with the natural forest type expected at that site. A degraded forest is a secondary forest that has lost, through human activities, the structure, function, species composition or productivity normally associated with a natural forest type expected on that site. Hence, a degraded forest delivers a reduced supply of goods and services from the given site and maintains only limited biological diversity. Biological diversity of degraded forests includes many non-tree components, which may dominate in the under-canopy vegetation. **Degradation** is … any combination of loss of soil fertility, absence of forest cover, lack of natural function, soil compaction, and salinization that either impedes or retards unassisted forest recovery through secondary succession. Reduction of forest cover, forest degradation and its fragmentation leads to forest biodiversity loss by reducing available habitat of forest-dependent species and indirectly through disruption of major ecological processes such as pollination, seed dispersal and gene flow. Forest fragmentation may also hamper the ability of plant and/or animal species to adapt to global warming as previously connected migration routes to cooler sites disappear. In certain forest types, fragmentation may also exacerbate the probability of forest fires, which further affects biological diversity in negative ways.
IPCC (2003a)	**Forest degradation** is a direct human-induced long-term loss (persisting for X years or more) of at least Y% of forest carbon stocks (and forest values) since time T and not qualifying as deforestation or an elected activity under Article 3.4 of the Kyoto Protocol.
IUFRO (Nieuwenhuis, 2000)	**Forest degradation** is damage to the chemical, biological and/or physical structure of a soil (soil degradation) and to the forest itself (forest degradation), as a result of incorrect use or management, and which, if not ameliorated, will reduce or destroy the production potential of a forest ecosystem (in perpetuity). Explanatory note: External factors, e.g. air pollution, can also contribute.

Forests provide a wide range of ecosystem services. For example, they protect soils from erosion; regulate the water regime; capture and store carbon; produce oxygen; provide freshwater and habitat; help to reduce fire risk (in the tropics); and produce wood and non-wood forest products (ITTO, 2002). Forest degradation, therefore, has the potential to adversely affect millions of people who depend, wholly or in part, on forest goods and services at a local scale, and billions of people who benefit from forest services at a regional or global scale.

WHY MEASURE IT?

Given their role in human well-being, the state of the forests is important to us all. We need to know if forests are being degraded and, if so, what the causes are, so that steps can be taken to arrest and reverse the process. Good information on forest condition and the extent of forest degradation will enable the prioritization of human and financial resources to prevent further degradation and to restore and rehabilitate degraded forests.

More specifically, information generated from the measurement of forest degradation can be used for:

- reporting to international conventions and processes on the status and quality of forest resources;
- the design and implementation of policies, programmes and forest-management measures to take preventive and corrective action through the restoration of degraded forests, the rehabilitation of degraded forest lands and sustainable forest management (SFM);
- the design and implementation of payment mechanisms or other incentive schemes for forest ecosystem services such as carbon offsets and conservation easements.

Countries are required to report information on the state of their forests, and their efforts to tackle forest degradation, at the international level. At the tenth Conference of the Parties to the Convention on Biological Diversity (CBD) in 2010, for example, the parties adopted the Strategic Plan for Biodiversity 2011–2020. This plan includes the Aichi Biodiversity Targets, among which is a target for the reduction of forest degradation. To determine if this target is reached, an effective process for monitoring and reporting on forest degradation is required.

The agreement to establish a mechanism under the United Nations Framework Convention on Climate Change (UNFCCC) aimed at a reduction in (greenhouse-gas) emissions from deforestation and forest degradation (REDD) provides another reason to measure forest degradation. REDD (and its more evolved and broader form, REDD+) has the potential to generate substantial funds for developing countries that would be used to reduce forest degradation (and therefore greenhouse-gas emissions) and to restore or otherwise improve the management of forests (thereby increasing forest-based carbon sequestration).

TOWARDS GUIDELINES

Under the umbrella of the 2010 Global Forest Resources Assessment (FAO, 2010), the Collaborative Partnership on Forests (CPF) and other partners initiated a study to identify the elements of forest degradation and the best practices for assessing them. That study built on existing processes and past initiatives relating to or requiring the measurement of forest degradation, including:

- the first of the four global objectives on forests agreed to by members of the United Nations Forum on Forests, which includes ‘increasing efforts to prevent forest degradation’;
- the Aichi Biodiversity Targets of the CBD, which included indicators on ecosystem fragmentation and connectivity, both of which are related to forest degradation;
- nine ecoregional processes on criteria and indicators (C&I) for SFM, which have been operational since 1992;
- three past meetings of experts to harmonize forest-related definitions, including one in 2002 that made a recommendation for a core definition of forest degradation;
- experiences in other sectors and through the CPF.

The technical work comprised three aspects:

- the development of a better understanding of the concept of forest degradation (see Annex 1 and FAO, 2009);
- the identification of suitable C&I for measuring forest degradation applicable to a range of multilateral processes;
- from these, the writing of chapters on assessing and measuring quantifiable aspects of forest degradation at the local, national and global levels.

The study involved:

- dispatch of questionnaires to FAO Global Forest Resources Assessment national correspondents and a survey of existing practices to establish what is being measured;
- the preparation of an annotated bibliography and an analytical study on definitions, which provided a framework for the process;
- case studies describing proven or promising methodologies and tools for assessing different aspects of forest degradation;
- technical meetings and discussions to review the results and recommend actions to improve the measurement, assessment and reporting of forest degradation;
- the preparation of initial guidance on how to measure different aspects of forest degradation – as represented by the present document.

Purpose of this document

This document is intended to provide relevant agencies and other stakeholders with direction on measuring forest degradation. It can be used for the development of programs for assessing forest degradation, and should be regarded as a precursor to the development of comprehensive globally applicable guidelines in the future.

Many factors affect the state of a forest and therefore have a bearing on forest degradation. Some of these, including policy and institutional settings, markets, trade and land tenure, fall outside the scope of this document, which focuses on measuring specific physical and biological effects and, in some cases, the direct causes of forest degradation.

The document has been prepared for use by those who work in the forest sector, particularly government officials, policymakers, forest managers and forest scientists. It is anticipated that it will be a living document that will be updated in the light of lessons learned in the assessment of forest degradation.

Elements of SFM

This document draws on the seven thematic elements of SFM, which are the extent of forest resources; forest biological diversity; forest health and vitality; productive functions of forest resources; protective functions of forest resources; socio-economic functions of forests; and legal, policy and institutional framework (United Nations, 2007). These seven elements, or variations of them, form the basis of all forest-related regional and global C&I processes (FAO, 2003). C&I were developed for monitoring and reporting on the status of forest management and progress towards SFM and can be applied at the national, subnational and forest management unit (FMU) levels. They also provide a suitable framework for the assessment of forest degradation (FAO, 2009).

In a broad sense, the seven elements of SFM encompass values placed on forest resources; therefore, forest degradation can be assessed in terms of the capacity of a forest to provide those values. A major difficulty in measuring forest degradation is the imprecise, multiple and often subjective interpretations of the concept (FAO, 2009). Any proposed method must account for and acknowledge the various perceptions of it.

Selection of key criteria

This document use four criteria derived from the seven thematic elements. They have been selected on the basis that indicators can be identified for each and that they are quantifiable. The four criteria are:

- forest biological diversity;
- biomass, growing stock and carbon;

- productive functions;
- protective functions.

For the SFM criterion of 'legal, policy, and institutional framework', indicators of forest degradation have proven less straightforward to identify. The relevance of the criterion 'extent of forest resources' was unclear; therefore, neither of these is addressed in this document. To some extent the criterion 'socio-economic functions of forests' is addressed by the criterion 'productive functions of forest resources' because a loss of forest productivity would have direct implications for many of the socio-economic benefits of forests. Indicators have been included, therefore, for forest goods. On the other hand, it was considered that the measurement of all forest services is too complex to be included in the document in its present form (see, for example, Box 1.2). While changes in the supply of forest goods can be measured directly, many services can be measured only indirectly. Biomass, growing stock and carbon are grouped in a chapter on biomass, recognizing that a measure of biomass can be obtained through the measurement of growing stock or directly through biomass measures, and measures of carbon stock can be derived from either. Chapter 6 presents indicators of soil erosion because the sustaining of forest goods and services requires stable and fertile soils. While other aspects of soil degradation, such as salinization, soil structure decline, organic matter loss, soil nutrient mining and contamination, are not considered here, ideally they would be included in subsequent iterations of this document.

Considerations

To be useful for a range of purposes, guidelines for assessing forest degradation need to address the following issues:

Flexibility. Guidelines should be sufficiently flexible to suit the circumstances of individual countries. For example, countries should be able to adapt international definitions of forest degradation to ensure that they are relevant in the national context, cost-effective and able to harness synergies with ongoing national processes. On the other hand, to enable global assessments and the use of national measures in international processes, comparability between countries is important. Therefore, a workable balance is needed.

Spatial scale. Forest degradation may need to be assessed at different scales to suit different purposes. For example, assessment at the scale of a stand or site is often needed for effective local-scale corrective action. On the other hand, assessment is needed at larger scales for national and international reporting and other purposes.

Temporal scale. What is the appropriate time scale within which to consider degradation? Short-term fluctuations in the capacity of a forest to produce certain goods and services may be part of a natural cycle or the result of planned human interventions (e.g. silvicultural treatment). Forest degradation may be more detectable on a longer time scale, and the range of 'acceptable' fluctuation may depend on the objectives of management.

BOX 1.2

Difficulties in measuring forest hydrological services

The regulation of water supply by forests may be considered to be an ecosystem service. Converting forest to non-forest increases dry-season flows, at least in the short term. Therefore, a process that might be considered to be forest degradation can actually improve an ecosystem service. On the other hand, the same process may diminish another ecosystem service because water quality is likely to be reduced. This is an example of how the definition of forest degradation can vary depending on perspective.

The difficulty of generalizing. Direct comparisons of the extent of degradation can only really be made between the same forest types and possibly only in terms of the function and state of the resource.

Thresholds. Degradation can be considered as both a state (i.e., the forest is either degraded or not degraded) and a process (where there may be thresholds along a continuum of degradation). Thresholds or reference states are needed to determine the state of a forest, or the extent of degradation along a continuum, and these may differ between and even within countries.

Forests change continuously due to natural processes and human intervention. However, when a parameter of forest condition passes beyond a certain threshold or amount, the forest may be said to be degraded. Similar to the concept of threshold is that of tipping point – the point at which a process of degradation becomes irreversible (without intervention), leading to a changed state (e.g. the conversion of forest to non-forest; Figure 1.1).

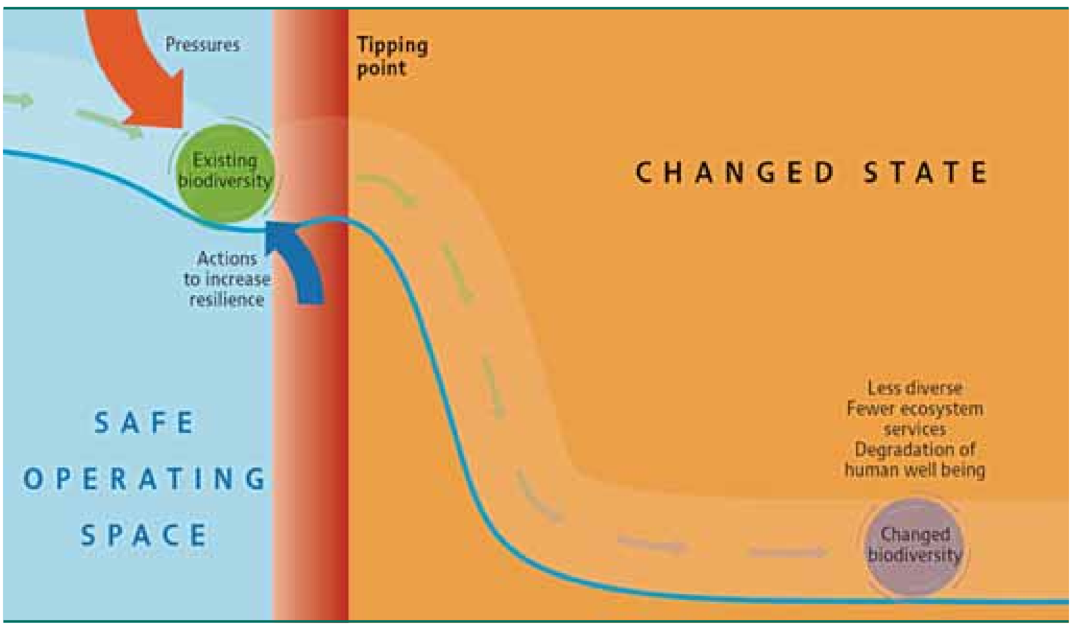
Resilience. Avoiding irreversible change – tipping points – may be more important than sustainability. This may require managing the resource in a way that enhances resilience – that is, the capacity of a forest to change without precipitating a radical shift in overall structure and function.

Differing perceptions. 'One person's degraded forest is another person's livelihood.' The objectives of management and use, as agreed by stakeholders, should be reconciled with the intended functions and state of the forest resource.

Causes of degradation may be human-induced or natural. It may be necessary to report separately on human-induced and natural causes of forest degradation.

Some of these issues are discussed in greater detail in Chapter 2.

FIGURE 1.1
Forest resilience and tipping points



Source: Secretariat of the Convention on Biological Diversity (2010).

Natural and human-induced degradation are often interdependent, since human actions can affect the vulnerability of a forest to degradation from natural causes (e.g. reduced stocking due to harvesting can lead to increased sensitivity to wind damage), while natural damage can also lead to increased human-induced disturbance (e.g. natural forest fire can lead to encroachment by shifting cultivators) and the deforestation of steeply sloping land can lead to widespread, severe erosion. Distinguishing between natural and human-induced causes may be difficult when abiotic and biotic factors are triggered by changes in weather patterns (perhaps as a result of human-induced climate change) that lead to a greater frequency, scale and impact of forest degradation.

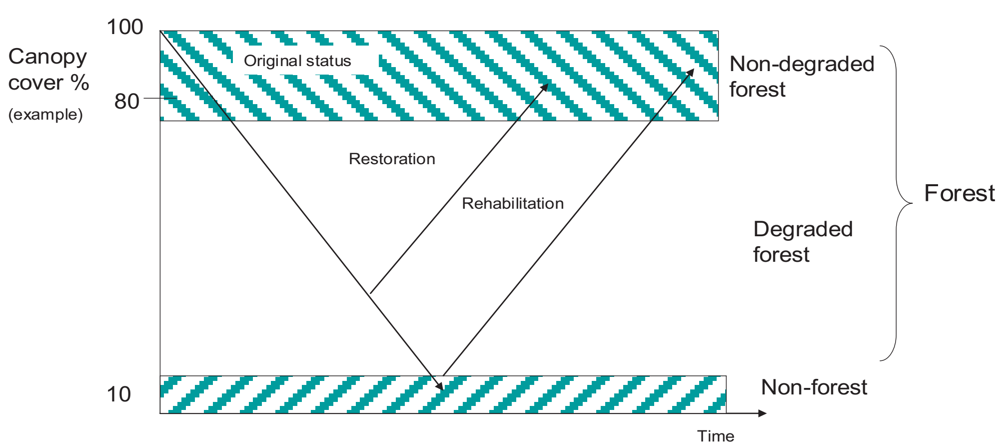
Forest degradation is usually associated with a reduction in vegetative cover, especially trees. There are exceptions, however, such as the 'empty forest syndrome' brought on by excessive hunting and/or the high-grading of commercially valuable timber species (FAO, 2009).

Degradation can be but is not necessarily a precursor to deforestation. Forests may remain degraded for a long time but never become completely deforested; change can also be abrupt, such as when an intact forest is converted to another use. At any point on the continuum depicted in Figure 1.2, forest degradation can be halted or reversed by forest improvement or other management interventions, including restoration through silvicultural measures and the rehabilitation of degraded non-forest through reforestation.

Data limitations. In the most recent Global Forest Resources Assessment (FAO, 2010), many countries were unable to report on a wide range of forest-related parameters. In some cases data have not been collected and in others they have not been processed. Data on forest degradation are likely to be even more difficult to obtain.

Baselines and reference states. The measurement of degradation requires the establishment of a reference state – a baseline or 'ideal state' – against which change can be assessed. Given that forests are always changing, and that forest condition is partly a matter of perspective, establishing a baseline is not an easy task.

FIGURE 1.2
Degradation thresholds



Note: A canopy cover of 100% refers to 100% of the average canopy for a given forest type.

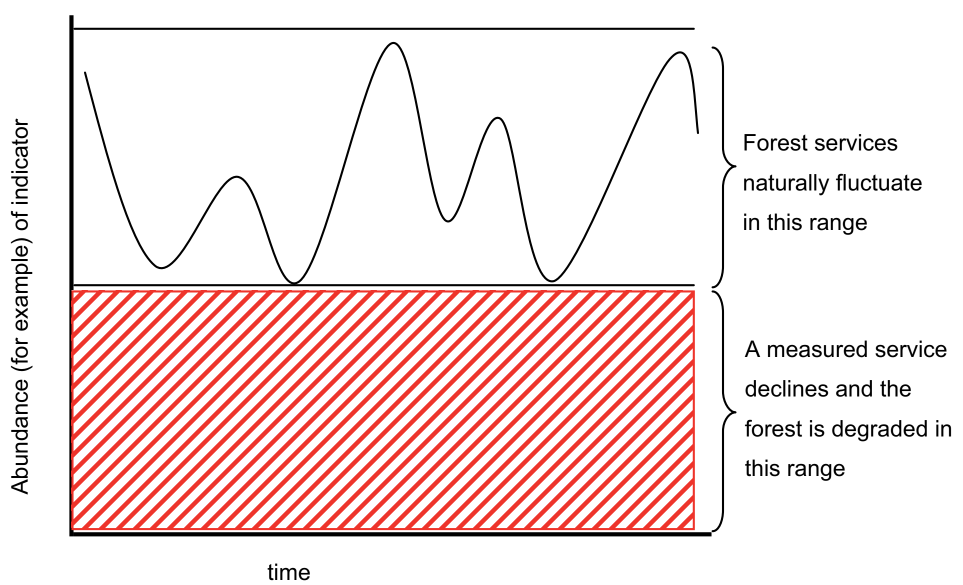
One starting point would be to use primary forests as a baseline, including the various successional stages. However, this seemingly reasonable approach is problematic because even sustainably managed forests may lack some species, processes, functions or structures normally found in a primary forest (and would therefore be rated as 'degraded' in comparison). Even if a sustainably managed natural forest was selected as the baseline condition, a challenge would be to define the threshold at which that forest loses its potential to fulfil its functions (i.e. provide all expected goods and services) and becomes unable to recover that capacity. Moreover, most plantations would be considered degraded according to the biodiversity criterion, as would any other forest that is managed for a selected set of goods and/or services. Degradation is relative, not absolute, and therefore should be classified along a continuum.

How much is too much? A certain amount of degradation may be deemed acceptable under various forest management scenarios. In some cases, the extent of degradation could be quantified based on an estimated percentage loss or decline of a good or service, and thresholds set. For example, it is possible to determine, through measurement, that a forest is producing 20 percent less wood than in the past, or that the population of a given species is 30 percent below a minimum baseline level. By measuring indicators over time it should be possible to show that degradation is occurring, but the acceptable level of decline in a product or service will ultimately be a political decision.

Natural variation versus degradation. In forest systems there is always an inherent range of natural variation (e.g. Keane *et al.*, 2009). Degradation occurs when the production of an identified good or service is consistently below an expected value and is outside the range of variation that would be expected naturally (Figure 1.3). The expected value might be a management objective, or it could be the known natural level based on experience and/or observation of a particular forest ecosystem. The range of natural variation can only be known through long-term research or monitoring, but even this range may vary over time in the face of change in external influences such as climate and so requires periodic monitoring.

FIGURE 1.3

Range of natural variation is used to provide a reference level for degradation



2. Issues in defining and assessing forest degradation

FAO (2009) surveyed countries on how they defined forest degradation, how they assessed it, and the indicators used in its assessment. Countries use various strategies to define forest degradation. For example, they variously:

- use international or regionally developed definitions;
- have developed their own definitions that may be recognized legally;
- use vegetation categories in classification, with degradation considered (usually implicitly) as a change from one class to another. The criteria applied are canopy cover and tree height;
- use indicators of forest degradation without specifically defining it. Some set threshold values to identify degraded areas, typically related to timber production, productivity or stocking level;
- do not have a definition of forest degradation but define associated terms such as secondary forest and degraded forest land.

Most countries focus on aspects of timber production (i.e. stocking level, productivity and biomass density). Argentina's definition is one of the most comprehensive, specifying the loss/reduction of biomass, structure, species composition, function, productivity and the capacity to provide goods and services.

Local conditions have a strong influence on the indicators used (explicitly or implicitly) when defining forest degradation or degraded forest. For example, in the Russian Federation focus is given to the sanitary condition of the forest and in Iceland the vigour of trees undergoing regeneration is the main indicator.

Forest management that involves harvesting, thinning or prescribed fire present challenges for the definition of degradation because they modify the state of the forest – albeit usually temporarily. In this case, the time period over which an indicator may be useful is critical. For example, is a lightly logged forest in the humid tropics degraded only until gaps refill, or must the stocking of harvested species return to pre-harvest levels? If the former, degradation is brief; if the latter, harvested stands could be degraded for decades.

INDICATORS OF FOREST DEGRADATION

One-third of countries responding to the survey of FAO (2009) reported that they did not have specific indicators for assessing forest degradation. The others listed a wide range of (possible) indicators but the extent to which these are used in practice is unclear. Indicators listed by one or more countries included aesthetic values; area affected by fire; disappearance of biodiversity/species; erosion; forest/canopy cover; fragmentation; occupancy/dominance of invasive/introduced species; presence of pioneer species/indicator species; soil fertility; soil properties; soil structure; species composition; stock density; production/value of timber and non-wood forest products (NWFPs); water quality; wildlife habitats; and wildlife risk.

Several of these indicators can be assessed on the basis of information already collected as part of national forest inventories or biodiversity assessments. Some (e.g. aesthetic values, soil properties, soil structure and wildlife risk) are difficult or costly to measure and probably remain areas for future work. All the identified indicators can be useful in assessing degradation, but they do not represent a systematic approach. The development of such a systematic approach should be undertaken according to the purpose of monitoring and the feasibility of measuring potential indicators.

Some countries apply a transition matrix to forest types, development classes or age categories as a basis for determining the process of change in forests and woodlands. This can provide a comprehensive approach to monitoring degradation based on information obtained from forest inventories.

There appears to be a general view that different indicator sets may be required for (natural/semi-natural) production forests; protected areas; and planted forests. In production forests, stocking density, age structure and species composition are typically used as indicators. Most respondents to the FAO (2009) survey recognized that both natural and human-induced degradation need assessment. Commonly, human intervention is held to cause degradation if it affects the functionality of forests; temporal changes such as thinnings or selective cutting *per se* are generally not to be considered degradation. Some countries, all with natural tropical forests, consider clear-cutting to be degradation (in one case including in plantations).

SFM elements as a framework for assessment of forest degradation

Views differ on the suitability of the C&I sets as frameworks to assess forest degradation. While, in general, C&I are considered an appropriate tool for this purpose, a number of respondents to the FAO (2009) survey expressed reservations because the C&I have been elaborated for broader purposes and many indicators are unsuitable for measuring the process or degree of degradation. (In this document, four criteria based on the seven common elements of SFM are used as a framework for indicators of forest degradation; see Chapter 1).

COMPARATIVE ANALYSIS OF DEFINITIONS RELATED TO FOREST DEGRADATION

There is strong commonality between the CBD, FAO and ITTO definitions of forest degradation (see Box 1.1) – all have a broad scope within the context of SFM. The Intergovernmental Panel on Climate Change (IPCC) approach is different because it is focused on a single element – forest carbon stock. National definitions demonstrate a mix of broad and narrow (productivity-focused) approaches.

Forest structure appears in four international definitions (those of CBD, FAO (2002b), ITTO and the International Union of Forest Research Organizations – IUFRO) and species composition in two (CBD and ITTO). However, structure is not defined in any of these definitions and could include any or all of various possible dimensions (e.g. age, tree diameter, tree size, development class and canopy structure). This is likely to need clarification in any global definition. In addition, the CBD definition includes forest function and ITTO's definition includes forest dynamics. The range of national definitions covers all these elements except forest dynamics and includes several that are not included in international definitions (e.g. stocking level, age structure, biomass density and sanitary condition).

The notions of supply capacity and forest goods and services appear in almost all international definitions – the exceptions are those of IPCC and IUFRO – and are also common in national definitions. The CBD and ITTO definitions refer to productivity and the former also to biodiversity. The restriction of the IPCC definition to forest carbon stock makes it different from all others, even though biomass density is a common element in many national definitions.

Ecosystem resilience and the degree of degradation are included explicitly only in the ITTO definition and are clearly operationally challenging. The CBD definition treats degraded and secondary forest as largely synonymous, while ITTO is more nuanced, separating degraded primary forest from secondary forest (where secondary forest is defined as forest that has regrown on land largely cleared of its original forest).

Most definitions, including those of the CBD and ITTO, specify human-induced causes of forest degradation. The FAO (2001, 2002b) definitions are exceptions; being comprehensive, they do not differentiate causes.

In both the CBD and ITTO definitions the reference state is natural forest. FAO (2002b) implies comparison with a previous state. National definitions give a range of reference states but a common interpretation appears to be 'what is expected on the site under similar conditions'.

The CBD, FAO (2001) and ITTO definitions specify the spatial scale at the stand or site level. The IPCC definition does not; the use of those scales may not be appropriate for the monitoring of carbon stock and both the CBD and ITTO definitions also recognize, in explanatory texts, the need for landscape-level assessment. In general, national definitions do not specify a spatial scale.

Temporal scale is long-term in the CBD, IPCC and ITTO definitions, but it is not clear what this means in practice. The IPCC definition states that degradation occurs over a specific (but unstated) period of time from a specific (but unstated) date. The difficulty in defining temporal scale led FAO (2001) and possibly also IUFRO to omit this element from the definition.

The exclusion of non-forest areas is generally implicit in definitions of forest degradation. The IPCC's framework definition, however, excludes deforestation and activities under Article 3.4 of the Kyoto Protocol. Planted forest is excluded from the ITTO definition (even though restoration and rehabilitation often involve tree-planting).

DISCUSSION

In general, the review of existing definitions in FAO (2009) shows that many are either very general or that their focus is on the reduction of productivity, biomass or biodiversity. All existing definitions of forest degradation are compatible with the generic common definition. The ITTO definition is probably the most comprehensive, and the CBD definition is quite close to it. The IPCC definition is narrower in scope, focusing on carbon.

Definitions that allude to multiple forest benefits may treat forest values in a comprehensive manner but are more difficult to use for international purposes in a consistent and transparent way. A definition on the basis of a loss of potential supply of 'goods and services' or 'benefits' requires subjective decisions in determining whether an area has been degraded. It is also subject to tradeoffs in which one good or service may be reduced or lost while another increases or is restored. Moreover, reductions in potential supplies of benefits can be achieved by legislation or regulation (for example, by restricting access to the services a forest can provide). Therefore, such definitions can imply that forests can be degraded (or the reverse) by rule, without any corresponding biological or physical changes (IPCC, 2003b).

The issue of thresholds between non-degraded forest, degraded forest and non-forest needs consideration, particularly in the context of REDD, which is a potentially powerful policy instrument that could have a significant impact on forest conservation. The higher the threshold between forest and non-forest, the earlier the degradation process becomes defined as deforestation. If the REDD mechanism were not to cover forest degradation processes, higher thresholds could be justified, as this would avoid allowing the degradation process to continue until it reaches the low threshold, thereby making the area eligible for REDD.

From a climate integrity point of view it is crucial to have a monitoring system for land-use-based emissions and sequestration that covers all relevant land uses independently of how they are classified or defined. In other words, although the threshold value between forest and non-forest may become an issue, the problem of defining forest degradation does not exist if the interest is in carbon stock change. If, on the other hand, the monitoring system does not cover all relevant land uses, the problem of forest definition and its thresholds becomes important and there will almost certainly be major leakage as countries tweak the threshold values between forest and non-forest lands in their national definitions of forest.

Most definitions of forest degradation refer to or imply application to natural forest, although planted forests may share the same criteria in some cases (e.g. if soil erosion is a key criterion, then planted and natural-forest soil changes would be comparable). In practice, the distinction between planted and natural forest is not always clear, since planted components are common in many modified natural forests.

Temporal change

The treatment of temporal changes in the forest is crucial for definitions of degradation. Reductions in crown cover or growing stock that cause short-term carbon emissions, such as selection, thinning or shelterwood cuttings, do not degrade a forest if properly designed and carried out (on the contrary, they can improve forest condition).

The issue of temporary change needs to be addressed in estimating changes in carbon stocks so that practices that cause short-term variations in carbon stock (e.g. selection cutting or thinning) are not considered, as a rule, to constitute degradation. In order to exclude short-term changes in the forest growing stock as part of SFM interventions, the ITTO, CBD and IPCC definitions have incorporated the ‘long-term’ element lacking in the FAO (2001) definition. However, none of the definitions specify what ‘long term’ means. While restricting forest degradation to situations exhibiting long-term effects is helpful in minimizing the effects of short-term variability and normal management, it requires that long-term effects be specified. Moreover, the operationalization of definitions may require an estimate of the length of time over which an observed change is likely to persist, which is a source of uncertainty (IPCC, 2003a).

Some stakeholders do not agree with the inclusion of the ‘long-term’ notion, insisting that any (including short-term) reduction in the growing stock should be considered degradation. This may have two possible motives: to ensure the use of REDD for conservation only (i.e. no timber harvesting); and a desire to tackle illegal logging. Such a one-sided approach would have a significant negative economic impact on the forest sectors of many developing countries.

Spatial scale

The ‘short-term’ view of carbon-related forest degradation derives from the perception that a forest stand is the basic unit of decision-making in conserving or enhancing forest carbon. However, forest management decisions are based on planning, which concerns a territorially designated unit that may be a holding, a forest estate or another type of FMU (e.g. watershed or landscape). These units typically consist of at least dozens of stands of different ages or with other structural differences. The mix of individual stands is under constant change as a result of biological processes and management interventions; carbon stock reduction may take place in some stands while, at the same time, there is an increase in carbon stock in other stands as a result of biological growth. It is the territorial entity for which management objectives are set that should be managed and assessed for such objectives as the supply of forest goods and services in combinations that are appropriate for the local conditions.

Human vs natural causes

There is a common perception that any compensation for ecosystem services, including reduced or avoided degradation, should be related to a change in (projected) human action. If the use of a definition of forest degradation requires the separation of human-induced causes (e.g. for carbon accounting under REDD), the human-induced aspect needs to be incorporated. There are, however, practical difficulties in separating human and natural (direct and indirect) causes, including those that are not within the forest sector, as many of them are interrelated. In developing countries, forest degradation is typically caused by human action but in developed countries the main causes are natural – both discrete events and slow, chronic degradation. FAO (2001) does not differentiate between human-induced and natural causes in its definition because of the difficulties associated with doing so.

Goods and services

The various international definitions of forest degradation (and improvement) leave open several issues related to the scope of goods and services, area involved, time scale, causes and possible threshold values. Therefore, operational definitions of forest degradation for specific purposes should, as appropriate:

- identify the goods and services to be assessed;
- provide a spatial context for assessment (land area identification);
- specify a reference state;
- cover both process and state (i.e. degradation/degraded forest);
- specify relevant threshold values;
- specify the reasons for degradation (human induced/natural) (when required by the definition);
- include an agreed set of variables;
- set out indicators (and their proxies if necessary) for measuring change.

Additional elements could be added or ignored, as required. It is important to initiate or expand assessment efforts independently of the eventual development of formal international-level definitions of forest degradation.

The use of proxies

Due to persistent data problems, the use of proxies (e.g. canopy-cover percentage) for indicators will continue, but only when it is clear that they provide relevant information on the degradation aspect of interest. It would be most cost-efficient if degradation could be established as a measurable sustained decrease in canopy cover (with canopy cover remaining greater than the minimum to qualify as forest). However, remote sensing methods need to be complemented by other methods (such as biometric field observations, biodiversity assessment and rapid rural assessments) to capture changes in forest values, goods and services and to fill data gaps.

Forest structure can be interpreted as an implicit reference to the growing stock, which may be used as a proxy for several purposes. A broader approach is likely to be necessary, however, and Lund (2009) proposes three commonly used proxy indicators:

- reduction in biomass for growing stock or carbon stored, which can be associated with a reduction of canopy cover and/or the number of trees per unit area;
- reduction in biological diversity, which can be associated with the occurrence of species (dominant and non-dominant) and habitats;
- reduction in soil, as indicated by soil cover, depth and fertility.

These may go some way towards constituting a comprehensive initial approach to the assessment of degradation. Relatively simple indicators would be needed for changes in forest structure that indicate degradation (and resilience) in different forest types.

Guidelines for climate-change mitigation

Finding an operationally feasible approach to forest degradation in the international climate regime is a pressing challenge. It is possible that stand-level-related definitional issues can largely be avoided if changes in carbon stocks are estimated across a designated forest area rather than at the stand level. A number of other issues still need to be addressed, however, including (UNFCCC, 2009b):

- how to deal with natural disturbances;
- how to distinguish between natural and non-natural disturbances and what the monitoring implications might be;
- whether it possible to reconstruct historical trends/rates with existing data.

The question of common or country-specific definitions needs to be considered, too. The use of common definitions would improve consistency and comparability between countries (FAO, 2002a). Using national definitions for forest and forest degradation would be consistent with current and earlier practices for the preparation of national greenhouse-gas inventories and would enable parties to the UNFCCC to include or exclude various elements in their approaches to estimating reduced emissions from forest degradation. However, relatively few countries have operational definitions of forest degradation.

OPTIONS FOR FUTURE ACTION

The following options for future action may be considered:

- maintain the holistic generic definition of forest degradation to provide a common framework for definitions developed for particular purposes;
- maintain the understanding that forest degradation can be further defined for various specific purposes and that different indicators can be used for its assessment;
- for each purpose identify what needs to be known and by whom, and the purpose for which the data should be used in order to develop appropriate indicators;
- recognize that, for international purposes, forest degradation needs to be assessed at a higher than stand or site level, which has implications for an international definition, while stand/site-level assessment is needed for local-level corrective action. This approach would focus on assessing the forest degradation (or improvement) process over time without the need to specify the temporal scale in the definition;
- allow scope for national interpretation of international definitions of forest degradation to ensure relevance and cost-efficiency and to harness synergies;
- improve existing definitions to provide greater clarity and to increase their consistency and compatibility;
- expand efforts to measure and assess forest degradation.

3. Growing stock and biomass

The systematic measurement of tree parameters in the field is fundamental to the assessment of growing stock and biomass. Other assessment methods, such as remote sensing surveys, can be complementary and can help to improve the precision of estimates involving large areas. To assess change in the growing stock and biomass, a monitoring system involving repeated, consistent measurements is required.

Change in growing stock and biomass can be an indicator of forest degradation, complemented by indicators of other aspects of forest condition. Long-term monitoring is required to take into account cyclical or temporary fluctuations. Multi-purpose forest monitoring and assessment are efficient systems for monitoring growing stock and biomass because they also provide information on the management objectives and multiple functions of forests and the multiple uses and users of forest and tree resources.

WHAT TO MEASURE

For effective forest management and planning it is vital to know the volume of wood and biomass resources held in forests at both the national and stand levels, as well as how much those resources are growing or declining. The wood volume of a forest is often referred to as the growing stock, which is a narrower concept than that of forest biomass (Box 3.1).

Estimates of growing stock are used to evaluate and monitor the commercial potential of a stand or forest for timber and fuelwood production. Information on growing stock is also essential for understanding the ecological dynamics and productive capacity of forests and for managing them within the limits of sustainable production. Estimates of the loss of or increase in growing stock over time can act as a quantitative indicator of forest degradation. Box 3.2 describes the estimation of growing stock in FAO's Global Forest Resources Assessment.

BOX 3.1

Defining growing stock and biomass stock

Growing stock – volume over bark of all living trees more than X cm in diameter at breast height. Includes the stem from ground level or stump height up to a top diameter of Y cm, and may also include branches up to a minimum diameter of W cm (FAO, 2006a).

Typically, this definition is applied as 'stem volume in forests of all living trees more than 10 cm diameter at breast height (or above buttresses if these are higher), over bark measured from stump to top of bole. Excludes: smaller branches, twigs, foliage, flowers, seeds and roots' (FAO, 2011). Commercial volume is derived from the total growing stock and constitutes the proportion of growing stock that is represented by commercial tree species, fulfilling minimum criteria for quality, dimension and sometimes age/development stage.

Biomass stock – organic material, both above-ground and below-ground and both living and dead (e.g. trees, crops, grasses, tree litter and roots) (FAO, 2006b).

BOX 3.2

Growing stock in the FAO Forest Resources Assessment

The Global Forest Resources Assessment 2010 (FAO 2010) provides estimates of growing stock per hectare at a national level and also information on the growing stock of each country's ten most common species. Estimates of total growing stock, by broad forest type, are usually derived from available inventory data; especially in the tropics, however, such data are often scarce or old. Estimates of growing stock may include all trees of all species, regardless of whether they are of commercial size or quality and regardless of whether they are growing in areas available for wood supply (although forest inventories are often limited to commercial species). Estimates of growing stock per hectare are derived by dividing total growing stock by forest area, sometimes by broad forest type. Comparative periodic data on the average growing stock estimates at the national level (as reported in national forest inventories and the FAO Forest Resources Assessment) can indicate trends in the quality of the forest resource but, on their own, they are insufficient for assessing forest degradation.

Forest biomass[1] is another measure of ecosystem productivity and also of the role of forests in the global carbon cycle. Although closely correlated to – and often estimated directly from – growing stock, forest biomass constitutes an important characteristic of forest ecosystems in its own right (FAO, 2008).

The assessment of growing stock and biomass densities relies primarily on field measurement; continuous and consistent measurements, usually in permanent plots, are needed to detect changes in growing stock over time. A negative trend – that is, a decline in the volume of the growing stock and biomass – can indicate forest degradation, but long-term monitoring is required to account for temporary fluctuations.

Data needs

To generate estimates of growing stock and biomass the following basic data are typically needed:

- sample data on the tree parameters used in models of growing stock and biomass (see examples in Annex 2);
- the size of the sample area;
- the total extent of the forest of interest.

Depending on the model used to calculate growing stock and biomass, tree parameters may include diameter, height/length, branch length, species, health status, increment rings and basal area.

The sample area may be predetermined (i.e. if a known number of fixed-size plots is used), or determined after field measurement (if plot size is not fixed). To produce estimates of growing-stock density and biomass density by land-use/cover class, field plots are classified according to a predefined land-use/cover classification system.[2] For a given plot, the dominant land-use/cover class can be assigned to a plot or to predefined plot sections, or the inventory team can subdivide plots into land-use/cover classes in the field. In the latter case, the size of each land-use/cover section is determined by measuring its length and width or by estimating the proportion of the plot occupied by each section (an approach typically used for circular plots).

[1] In this chapter, 'forest biomass' mainly refers to above-ground living biomass.

[2] The land-use/cover classification system, as defined by FAO, is a combination of a land-use classification system and a land-cover classification system. 'Forest' is a land-use class, but the different forest types are usually distinguished by different land-cover classes.

The total area of the forest of interest can also be subdivided into land-use/cover classes, either by extrapolating from land-use/cover class distribution within the plot or through a more comprehensive remote sensing survey. The latter is usually more accurate, especially for classes that are less frequent in the landscape.

Data that help in understanding growing-stock and biomass development and dynamics should be collected. Parameters of interest include those related to trends in land-use/cover class changes; land-use designation/protection status; regeneration (e.g. stem density, by species); the frequency of tree stumps (e.g. species, diameter at stump height, and years since harvest); tree harvest/extraction (e.g. frequency, change trend, trend reason, users' expected/desired future tree cover, and timber exploitation system); silvicultural system; tree products/services (e.g. species and supply/demand trends); tree canopy cover; the existence and quality of management plans/agreements; environmental issues; stand history; soil (e.g. productivity, drainage, nutrients and pH); fire (e.g. type, area and frequency); local people (e.g. population size, dynamics, economic pursuits and settlement history); and proximity to infrastructure and accessibility (e.g. all-weather/seasonal roads, markets and settlements).

MEASUREMENT METHODS

To some extent the availability of financial resources and human capacity determines the method for assessing growing stock and biomass. In a best-case scenario, multi-purpose field-based forest inventories collect primary data on tree species, diameter and height, land use, and so on. In most developed countries, field-based national forest inventories are undertaken on a reasonably regular and frequent basis, but this has not always been the case in developing countries. In the absence of timely, broad-based inventories, partial inventory data and/or extrapolations from past inventories must suffice for volume or biomass calculations. Combining permanent sample plot methodology with a remote sensing survey can also be used to assess forest cover and other parameters at a relatively moderate cost.

Field survey

In a field survey, forest inventory teams collect data on the ground. For a relatively small forest area, such as a logging coupe, it is possible (and often required) to conduct 100 percent inventories (also called full-cover or wall-to-wall inventories) in which all trees in the stand (usually above a specified minimum diameter) are measured. For larger-scale inventories, such as at the landscape, provincial or national level, a 100 percent inventory is likely to be impractical and prohibitively expensive. A sampling strategy is therefore required whereby measurements are made in permanent and/or temporary sampling units, and those measurements are used subsequently to estimate values for the entire forest area. The sample area is the total area of all sampling units in which measurements are made.

The sampling procedure can be random or systematic. It is usually more efficient to apply a systematic sampling method, as this tends to provide the best representation of the distribution of land uses and forest types. Sampling can also be pre-stratified in order to intensify the sample in strata that are more heterogeneous or of higher priority, thus increasing the precision of estimates where it is most required. The sampling units may be stands, plots, strips or points, and plots may be circular, rectangular or square (or some other shape) and of fixed or variable size.

Plot size is determined according to the expected number of measurements of the parameters of interest. For example, plots for measuring small trees can sometimes be smaller than plots for measuring larger trees, as the density of small trees is often higher than it is for larger trees (and therefore a similar number of stems can be measured in a smaller area). The number of plots is determined by the need for statistical precision, especially for key parameters, and by cost and time constraints. More time and effort is usually required to measure a widely dispersed set of plots than plots arranged in clusters (i.e. ‘cluster sampling’).

Remote sensing survey

A remote sensing survey (e.g. using aerial photos or satellite images such as those generated by LandSat or Advanced Space-borne Thermal Emission and Reflection Radiometer – ASTER) can be used for either full-cover or sampling approaches. In a sample-based approach, observations are made in sampling units (sample area), while in a full-cover approach the entire area of interest (e.g. a landscape, province or nation) is measured. Remote sensing observations can be used in particular to determine the extent or area of land-cover (or land-use) classes. This can greatly assist in extrapolating volume and biomass densities generated by field-based measurement over large areas and over time in repeated assessments to estimate changes in volume and biomass, or to stratify the design of field-based sampling.

Radar and laser-derived space-borne or air-borne remote sensing can also be used to capture data for estimating both volume and biomass stocks, but their accuracy depends on the ability to calibrate and validate measurements with field-based data. These technologies are still expensive and experimental but show promise, particularly for areas that are difficult to access in the field.

Equipment and data collection

A variety of instruments and tools is available for measuring tree parameters, depending on budget and expertise. Table 3.1 sets out some of the basic tools that can be used for direct measurement and observation.

Typically, the tree stem diameter is measured over bark at a height of 1.3 m above the ground, commonly referred to as breast height (hence the expression diameter at breast height – dbh).

TABLE 3.1

Basic tools for measuring growing stock

Main tree parameters of interest	Equipment
Stem dbh	Diameter tape
	Scale stick
	Caliper
Tree height/branch length	Clinometer
	Altimeter
	Stick/ruler + measuring tape
	Rangefinder
	Hypsometer
	Relascope
Species identification	Botanical field guide and/or local knowledge
	Plant press for sample collection
Geographic location	Global positioning systems
	Compass + measuring tape
	Topographic maps

Tree height is the vertical height of a tree from the ground to the top of the tree. Commercial bole height is the length of the bole from the stump to the height of the bole at the point of smallest merchantable diameter (i.e. the minimum bole diameter able to be used as timber). Both total height and commercial bole height can be estimated using a variety of tools. The method using a scale stick, for example, is as follows: stand a known distance from a tree with the entire length of the tree visible; hold the scale stick a set distance from the eyes and align the base of the stick with the base of the tree; measure the height (either total height or merchantable height) of the tree against the scale of the stick; use trigonometry to calculate the real height of the tree. A more accurate (and expensive) tool is the clinometer, which measures angles and enables the user to determine the height of a tree when standing at a given distance. Laser rangefinders can also be used, ideally with leaf filters to reduce false readings; these are expensive but also very quick.

The measurement of tree height is particularly difficult in forests where tree tops are not visible because of the dense canopy (e.g. in closed tropical forests). Given that it is far more costly to measure tree height than dbh, tree height is often measured for only a subsample of trees. On the basis of such a subsample the relationship between dbh and tree height can be modelled and applied to predict the height of all trees in the sample.

Tree shape and therefore volume vary between species and often within species. Species-specific volume and biomass models and/or wood densities, where they exist, should therefore be used. The identification of species can be a highly specialized skill, especially in the tropics (because of the great diversity of tropical forests). Local knowledge is essential, aided by field guides and, where necessary and available, herbarium specimens. Global positioning systems, maps, compasses and measuring tapes can be used to measure the location of plots and of trees within plots.

Data recording, validation and storage

During a field survey, data are recorded on field forms, which may be paper-based or digital (the latter involving the use of a digital data collector). Codes are often used to record data on the basis of predefined options, although this is impractical for some continuous values (e.g. dbh) and qualitative descriptions. It is strongly recommended that, as much as possible, data are recorded using well-defined recording units or codes, because descriptive data recorded as free text are laborious to analyse using numerical methods.

There are several advantages to using digital data collectors in the field over paper-based methods, including the following:

- Validation criteria can be programmed to provide inventory teams with immediate feedback on data that are inconsistent, thereby reducing human error in data recording.
- Digital data collectors can receive and store measurements made using digital instruments (e.g. laser rangefinders), reducing double-handling by operators.
- Data can be transferred directly to the main database without additional manual data entry.

A disadvantage of digital data collectors is the potential for a significant loss of data should the device malfunction, although this risk can be mitigated by frequent back-up. Another disadvantage is that the 'human' validation of data is often omitted, especially when readings are transferred electronically from an instrument to a digital data collector without human intervention. Moreover, digital instruments are relatively expensive and their maintenance requires considerable expertise, which may be unavailable in some countries, especially in remote areas. Where a paper-based system is used to record field data, data should be validated by supervisors before they are entered into the database. Once data have been collected, validated and entered into the database, further validation and the 'cleaning' of data (i.e. the detection and correction of incorrect values) should be undertaken.

Field data collection should also be verified using a rigorous quality-control system. This usually involves the re-measurement of a sample of field plots as an independent check of the original data.

The database is the core of the information system; it holds and safeguards forest inventory data and all information related to the inventory. Therefore, the sustainability of the monitoring system depends on the adequate maintenance of the database. Forest data, field instructions, nomenclatures, definitions, applied data processing procedures and models should all be recorded in the database to ensure the compatibility and comparability of successive inventories and monitoring programs.

Estimating growing stock and biomass

Tree allometry is the use of equations, models and functions to describe the quantitative relationship between various tree parameters. In combination with tree inventory data, allometric equations can be used to estimate tree volume (Box 3.3), tree biomass (Box 3.4) and, ultimately, the growing stock and biomass of forests at various scales. However, allometric equations and functions are lacking for many species and forest types, in which case more general models must be used (Box 3.5). In many countries, considerable research is required to improve the accuracy of species-specific and forest-type-specific estimates of growing stock and biomass.

BOX 3.3

Estimating tree volume

A commonly used and general equation to estimate the stem volume of an individual tree using field data is the following:

$$volume = dbh^2 / 4 \times \pi \times h_{tot} \times f_{form}$$

Where: h_{tot} = *tree height (expressed in m)*

f_{form} = *stem form factor*

$\boldsymbol{\pi}$ = *3.141596.*

Note that, in this equation, volume is expressed in m^3 and dbh in m.

The stem form factor is defined as the comparative volume (as determined by dbh) of the tree stem at specific heights and relates form and volume. The value of the form factor typically ranges from 0.3–0.8, depending on the shape of the tree.

When species-specific tree stem form factors are unavailable, a less accurate country-specific default value may be used which generalizes the form of all species contained in the inventory. Growing stock can be estimated on the basis of the volume of individual trees by extrapolating from estimates made in sample plots to the total forest area.

Example

The stem volume of a tree with dbh 18 cm, total height 9 m and the stem form factor 0.55 can be calculated using the above equation: i.e. volume = $(0.18)^2 / 4 \times 3.141596 \times 9 \times 0.55 = 0.126\ m^3$.

BOX 3.4

Estimating above-ground tree biomass

The following example shows an equation for above-ground tree biomass where a biomass function has been developed for the pantropics (Chave *et al.*, 2001).

AGB = exp[-2 + 2.42 ln(dbh)]

Where dbh is expressed in cm and above-ground biomass (AGB) in kg.

Example

The AGB of a tree with dbh 18 cm can be calculated using the above equation: i.e. AGB = exp(-2 + 2.42 × ln18). Since Ln(18) is 2.89, exp(-2 + 2.42 × 2.89) = exp(4.99) = 148 kg.

In this equation, AGB is estimated on the basis of dbh. Other biomass equations, however, may involve two or even three parameters (e.g. dbh, height and wood density). Note that all allometric equations work within certain thresholds; fore example the above equation holds for trees with minimal dbh of 10 cm.

BOX 3.5

Forest biomass calculation

To estimate biomass, the choice of method is determined by the availability of data and biomass estimation methods. FAO (2008) identified the following options in descending order of precision:

- where available, the use of country-specific functions for directly estimating biomass from forest inventory data, or country-specific factors for estimating biomass from estimates of growing stock;
- the use of other biomass functions and/or conversion factors (e.g. those developed for similar species and forests) considered to provide greater accuracy than the default regional/biome-specific conversion factors published by the IPCC;
- The use of IPCC default factors and values (IPCC, 2006), which continue to improve and are now available for various geographical regions and ecological zones.

Aggregating estimates

Volume or biomass estimates for individual trees in the sample area are aggregated to derive the total inventoried tree volume in the sample area. The growing-stock density (m^3 per hectare) and biomass density (tonnes per hectare) can be calculated by dividing the total inventoried tree volume or biomass by the sample area. Some volume functions with measurements at the stand level (e.g. basal area) generate estimates of volume density directly.

To calculate the total growing stock and biomass stock, these densities are extrapolated to the total forest area. An effort should be made to estimate sampling error and to provide a margin of error to account for sampling error, measurement error and other uncertainties in the design and implementation of the methodology. The only error that can be estimated using statistical methods is sampling error, but the uncertainty associated with other sources of error is often larger. Therefore, attention should be paid to quality control and quality assurance in order to minimize overall error (FAO, 1973).

Estimating changes in growing stock and biomass

To measure change in growing stock, repeated measurements of permanent sample plots can be made and differences between successive inventories assessed. Differences in volume and biomass over time are referred to as ‘growing stock change’ and ‘biomass change’, respectively. Among other things such measures allow managers and planners to better predict forest stocks from year to year.

$$\Delta\, GS_{t2\text{-}t1} = GS_{t2} - GS_{t1}$$
$$\Delta\, B_{t2\text{-}t1} = B_{t2} - B_{t1}$$

Where
$\Delta\, GS_{t2\text{-}t1}$ = *change in the growing stock between measurements taken at time 1 and time 2*
GS_{t1} = *growing stock at time 1*
GS_{t2} = *growing stock at time 2.*
$\Delta\, B_{t2\text{-}t1}$ = *change in the biomass between measurements taken at time 1 and time 2*
B_{t1} = *biomass at time 1*
B_{t2} = *biomass at time 2.*

When field data on volume and biomass are lacking for assessments of change (i.e. historical field measurements are missing), but temporal remote sensing data series exist, change estimates can be made by applying ‘static’ values of different biomass density classes to the changes in area in the various remote sensing biomass strata. This approach assumes that biomass densities don’t change over time within the remote sensing biomass strata, an assumption that may not hold in all cases.

MEASUREMENT FREQUENCY AND REPORTING

A rule of thumb for the interval between assessments of growing stock and biomass at the provincial or national scale is 5–10 years, but this can vary according to the required precision of estimates, the rate of ecological change, information needs and, above all, the availability of resources. A continuous (or annual) inventory approach can also be adopted by measuring, for example, 20 percent of plots per year to create a five-year cycle.

To ensure that measurements can be repeated, accurate information on the location of plots, and trees within plots, should be recorded. For reporting purposes it is necessary to document the threshold values used to estimate growing stock and biomass (e.g. all trees with a dbh ≥ 10 cm). These values are needed to harmonize data between inventories or countries and for global reporting.

ISSUES AND CHALLENGES

The biggest challenge in estimating changes in growing-stock volume and biomass is the lack of governmental programmes for the consistent monitoring of forest resources. Most countries have conducted certain forest inventories in the past, but few have carried out complete forest inventories at a national level and even fewer have followed up such inventories with consistent monitoring – as priorities and capacities have changed over time. In order to know the status of and trends in the national forest resource it is important to develop institutional capacity and knowledge in the assessment and analysis of forest data.

To produce high-quality estimates of growing stock and biomass, and changes to these, a robust forest monitoring system needs to be complemented by access to applicable models for calculating estimates – a significant stumbling block in many countries. National species-specific and forest-type specific models for volume and biomass estimates often do not exist or are incomplete; this is the other big challenge for improving estimates of growing stock and biomass.

Support from international organizations and the bilateral provision of expertise can play substantial roles in developing national capacities in national forest monitoring systems and the development of country-specific models. Such support should be provided consistently to enable the development of lasting in-country capacity.

4. Biodiversity

While it is possible to devise a single 'biodiversity score' or cumulative index, such a composite index may lack the sensitivity to detect the degradation of biodiversity. It may be too strongly influenced by one variable, for example, despite a decline in others. Ideally, therefore, the loss of biodiversity should be assessed using several indicators, in part because biodiversity refers to life at several scales. These indicators would be scored against expected benchmarks (for example, the abundance of certain species, the number of populations of functional species, or the number of ecosystem types) to determine the extent of degradation for each indicator and for the forest stand or landscape.

At the very least, an indicator requires measurements at two points in time or against a control value. The biodiversity indicators in this chapter should form a common set that could be employed to determine the *amount of degradation* in a local forest, regardless of forest type.

Degradation differs from forest loss, but a loss of forest in a certain portion of a landscape may have an overall degrading effect on landscape biodiversity. For example, Andren (1994) suggested that there was a threshold of 30 to 40 percent forest loss across a landscape, beyond which there would be non-linear declines in the occurrence of species. This threshold value has since been tested for various species and landscapes but, overall, it seems that generality is difficult and thresholds depend on the species of interest and forest type (e.g. Betts and Villard, 2008). Hence, thresholds may need to be determined based on the expected range of variation in an ecosystem, a community or a species of interest. Biodiversity thresholds and baseline information are critical for determining the usefulness of an indicator and represent a significant challenge for managers and researchers.

Biodiversity provides important ecosystem services, such as pollination (e.g. by bats, birds and insects), decomposition (e.g. by soil arthropods, fungi and micro-organisms), seed dispersal (e.g. by insects, birds, mammals and fish), resilience and disease reduction. The provision of goods and services – such as bushmeat or fibre – may also depend on the abundance of certain species.

Biodiversity indicators of forest degradation should be assessed at two scales: landscapes (multiple stands), and stands (individual groups of trees distinguishable from other surrounding groups of trees by their species composition). Both scales are important and require a different, but sometimes overlapping, set of indicators. In many cases, scaling up from stand to landscape will be required for reporting on degradation. Indicators should be amenable to data collection and easily repeatable, especially in countries with limited scientific and monitoring resources. They must be unambiguous and provide quantitative data that can be used to assess trends over time.

WHAT AND HOW TO MEASURE

Biodiversity indicators in other forest-related processes

Table 4.1 sets out indicators used for monitoring biodiversity in selected SFM C&I processes or forums. Not all of these are helpful for measuring degradation. Where they are, however, using them for the assessment of both SFM and forest degradation is likely to lead to efficiencies in monitoring, assessment and reporting.

TABLE 4.1

Biodiversity indicators of SFM, from five indicator sets or processes

Process or forum	Landscape	Ecosystem	Species	Genetic	Health	Other
ITTO	5.1: Forest protected area		5.4: Number of endangered, rare and threatened species* 5.6: Measures in place to protect listed species, species of interest, keystone species and seed trees*	5.5: Measures for protection of genetic diversity of commercial species or listed species		
Montreal Process	1.1c: Fragmentation of forests*	1.1a: Area and percentage of forest by forest ecosystem type, successional stage, age class and forest ownership or tenure* 1.1b: Area and percentage of forest in protected areas by forest ecosystem type, and by age class or successional stage	1.2a: Number of native forest-associated species 1.2b: Number and status of native forest-associated species at risk, as determined by legislation or scientific assessment* 1.2c: Status of on-site and off-site efforts focused on the conservation of species diversity	1.3a: Number and geographic distribution of forest-associated species at risk of losing genetic variation and locally adapted genotypes* 1.3b: Population levels of selected representative forest-associated species to describe genetic diversity* 1.3c: Status of on-site and off-site efforts focused on the conservation of genetic diversity	3a: Area and percentage of forest affected by biotic processes and agents (e.g. disease, insects and invasive alien species) beyond reference conditions* 3b: Area and percentage of forest affected by abiotic agents (e.g. fire, storm and land clearance) beyond reference conditions*	2a: Area and percentage of forest land and net area of forest land available for wood production
CBD	7.1: Patch size distribution, connectivity and fragmentation* 7.2: Area burned* 5.1: Change in forest area*	1.1: Percentage area of forest protected, by forest type 1.2: Percentage of threatened or vulnerable ecosystems protected* 5.2: Forest areas by class: primary, modified natural, semi-natural and plantation*	2.1: Change in the abundance of populations of selected species* 2.2: Change in the distribution of selected species 2.3: Number of species listed in the IUCN red list of threatened species, by category* 2.4: Changes in the status of individual species listed in the IUCN red list of threatened species *	3.1: Area managed for *ex situ* conservation of forest genetic resources 3.2: Area managed for *in situ* conservation of forest genetic resources	6.1: Number of invasive alien species in forests* 6.2: Number of invasive alien species controlled 6.3 Area of forest affected by invasive alien species* 7.2 Area burned*	4.1: Percentage of forest area under management that is certified* 5.3 Area of degraded forest*
Biodiversity Indicators Partnership	9.3: Forest fragmentation*	1.1: Extent of forests and forest types* 1.2: Extent of selected habitats	2.1: Living planet index 2.2: Global wild bird indicator 4.1: Change in the status of species listed in the IUCN red list of threatened species*	5.1: *Ex situ* collections	8.2: Number of and trends in invasive alien species*	6.1: Area managed and certified 6.2: Area managed that has been degraded and deforested

Forest Europe Indicators	4.7: Landscape-level spatial pattern of forest cover*	4.3: Area of forest and other wooded land classified as 'undisturbed by man', 'semi-natural' or 'plantations', each by forest type* 4.4: Area of forest and other wooded land dominated by introduced tree species* 4.5: Volume of standing dead wood and lying dead wood in forests and other wooded land classified by forest type* 4.9: Area of forest and other wooded land protected to conserve biodiversity, landscapes and specific natural elements, according to Forest Europe assessment guidelines*	4.8: Number of threatened forest species, classified according to categories used in the IUCN red list of threatened species in relation to total number of forest species*	4.6: Area managed for conservation and utilization of forest tree genetic resources (*in situ* and *ex situ* gene conservation) and area managed for seed production*

* Denotes indicators relevant to forest degradation.

Indicators

The biodiversity indicators set out in Table 4.2 apply to all forest types and to managed, used but unmanaged, and primary forests. Indicators were selected on the basis that:

- they should be sufficiently generic to apply globally;
- techniques should be available to allow measurement;
- there should be existing data sources;
- they should have the potential to be scaled up;
- a change in the indicator should indicate a change in biodiversity;
- the relationship between the indicator and the values or services of interest should be clear and understood easily by the decision-makers expected to make use of it.

Not all the indicators in Table 4.2 meet all these criteria, but they all meet at least some of them.

Satellite or other remote imagery may be useful for some indicators of forest degradation but only a few of those are associated directly with biodiversity. Ground-based biodiversity indicators can be viewed as complements to remotely sensed degradation indicators and may help to identify degradation in areas otherwise reported as 'not degraded' or 'possibly degraded' on the basis of remote sensing. Data collection for ground-based biodiversity-related indicators is more difficult and labour-intensive but is necessary to obtain a full understanding of forest condition. It should be undertaken on the basis of stratified samples of each forest type to at least the level of sub-biomes.

The minimum set of indicators that should be used to assess biodiversity-related forest degradation comprises 'ecosystem state' and 'forest fragmentation', both of which can be determined through remote sensing. Four types of species indicator (see Box 4.1 for a definition), which usually require some level of ground survey, are considered in detail here: tree community structure, focal species (listed, flagship and indicator), functional species, and invasive alien species.

A range of species are used commonly as indicators in forest management planning and for monitoring the impacts and effectiveness of forest management regimes (e.g. Oliver and Beattie, 1996; Noss, 1999; Azevedo-Ramos, de Carvalho and Nasi, 2010; Lewandowski, Noss and Parsons, 2010). Noss (1999) suggested the following as useful groupings:

- Area-limited species: species that require large areas of contiguous forest to maintain viable populations. These species typically have large home ranges (e.g. woodland caribou in Canada) and/or low population densities (e.g. many mammalian carnivores).

TABLE 4.2

Possible biodiversity indicators of forest degradation

Data collection method	Indicator	Measurement method	Relevant case studies or data source	Scale of measurement
Remote sensing	Ecosystem state (resilience)	Satellite or aerial photographs: expected forest type for climate elevation, soil and moisture condition	Surrounding area, protected areas, etc.	Stand or landscape
	Fragmentation/ intactness and road density	Satellite or aerial photos: area deforested, roads per km^2	United Nations Environment Programme-World Conservation Monitoring Centre, World Resources Institute	Landscape
	Ecosystem diversity	Satellite or aerial photography: extent of each ecosystem type	National forest inventories	Landscape (stand)
Ground-based (species-based) indicators	Expected community composition by forest tree species for the ecosystem type	Ground plots: species composition	Individual research, government survey, expert opinion, IUCN red list of threatened species	Stand and landscape
	Key indicator species, including threatened species, old-growth forest species, and hunted species*	Surveys for change in population size (relative or absolute)	IUCN red list of threatened species, local data on populations, expert opinion	Stand, landscape
	Invasive alien species	Remote sensing or ground-based surveys: area of forest affected	Local data	Stand, landscape
	Functional species	Surveys for change in population size, surveys for expected function products (e.g. fruit production)	Local data	Stand

* Hunted species (for bushmeat) are addressed in Chapter 5.

BOX 4.1

Definition of indicator species

Organisms whose presence is used to mirror environmental conditions or biological phenomena too difficult, inconvenient or expensive to measure directly. They should be sensitive to changes in the real phenomena of interest and should be used only when direct measurement is impossible or infeasible (Rolstad *et al.*, 2002).

- Dispersal-limited species: species that are limited in their ability to move between habitat patches or that face a high risk of mortality in trying to do so. These species require habitat patches in close proximity to one another, movement corridors, or crossings across barriers such as roads. Forest species in this category include flightless insects limited to forest interiors, small forest mammals, and large mammals subject to illegal hunting.
- Resource-limited species: species requiring specific resources (e.g. large standing dead trees, nectar sources or fruits) that are often or at least sometimes in limited supply. The number of individuals that a forest can support is determined by the carrying capacity at the time at which the critical resource is most limited. Species in this category include hummingbirds, frugivorous birds and cavity-nesting birds and many mammals.
- Process-limited species: species sensitive to the level, rate, spatial characteristics or timing of an ecological process such as flooding, fire, wind transport of sediments, grazing, competition with exotics, or predation. Species in this category include plants that require fire for germination or to reduce competition.
- Functional species and keystone species: functional species are species that are disproportionately responsible for key ecosystem functions. Keystone species are functional species that are also ecologically pivotal species, meaning that their impact on a community or ecosystem is disproportionately large given their abundance. Examples in forests include tree species that store most carbon, cavity-excavating birds, and herbivorous insects with large population fluctuations.
- Narrow endemic species: species restricted to a small geographic range, often at low abundances within that range (e.g. some herbaceous plants and large mammals).
- Special cases: species important in the forest ecoregion that do not fall within one of the above categories. This group includes disjunct or peripheral populations that are genetically distinct, and 'flagship species' that promote public support for more general conservation efforts.

The major criterion for the indicators proposed below is that relatively little investment in measurement could provide information about other species or processes as well. It follows, therefore, that:

- their measurement should be low-cost relative to available resources and the magnitude of the process(es) of interest (i.e. they have the quality of efficiency);
- they should respond rapidly and measurably to change in the conditions of interest (i.e. they have the quality of sensitivity);
- changes in the indicator should provide a disproportionately large amount of information about the status or change in status of other forest attributes correlated with the process being monitored (i.e. they have the quality of surrogacy);
- the species should be especially important for the local area (e.g. for culture, functional role, food or tourism).

For monitoring the multiple processes that together comprise degradation, an approach similar to the 'focal species' approach is required, with different species used to monitor different processes. In general, the degree to which the sensitivity of a single or a few species is correlated with other species or processes is poorly understood. Correlations are usually assumed (albeit based on sound reasoning) rather than proven (Lindenmayer *et al*., 2002). The data requirements for the use of species as indicators may be limiting, both to demonstrate the correlation between species abundance and degradation and to detect changes in populations with statistically valid confidence. Therefore, the use of species as indicators requires an understanding of the limitations of the technique, careful species selection, and considerable testing.

For most species-based indicators, the main parameter to be measured is change in abundance, although presence/absence may also be appropriate. In certain circumstances other measures may be used, for example the chemical composition of lichens if sulphur-based pollution is an issue. An indicator indicates but it does not necessarily show beyond doubt that there has been a change and it does not explain why the change has occurred. Often, a change in a species-based indicator will show where further investigation is required so that a management response to the change can be formulated.

Ecosystem state

'Ecosystem state' refers to the composition and structure of an ecosystem relative to the ecosystem predicted to occupy a given site in the absence of atypical disturbance or environmental change. Hence, this indicator can be written as 'area of forest that has changed state from that predicted for that site'. Key parameters are the dominant floristic (tree) composition and stand structure that may be expected for a given stand.

Resilience is the capacity of natural systems to self-repair following major disturbances, and the loss of biodiversity will often mean a reduction in that capacity (e.g. Thompson *et al.*, 2009). A decline in resilience may be caused by the loss of functional species groups resulting from environmental shifts such as climate change, or from a sufficiently large or continual alteration of the natural disturbance regime (Folke *et al.*, 2004).

Some changes in the relative abundance of dominant species may occur following disturbance with few apparent consequences for the ecosystem. In some cases, a forest may maintain its capacity to provide certain (most or all) ecosystem goods and services, even if its species' composition and/or structure are altered permanently. This resilience (Gunderson, 2000; Walker *et al.*, 2004) is strongly dependent on biodiversity (Balvanera *et al.*, 2006; Thompson *et al.*, 2009).

A negative change in state refers to a loss of resilience that leads to a shift from one ecosystem type to another, with a consequent change, and reduction, in certain goods and services. For example, if a forest is expected to be composed of mixed tree species but instead is mostly uniform, or it should have a closed canopy but is actually open or savannah, then the state has changed.[3] A relatively simple index of forest degradation on the basis of biodiversity change, then, could be the sum of the area of atypical or unexpected forest types on a given landscape, such as total area of open-canopy forest in a closed-canopy landscape. These changes are relative to the forest predicted for a given site or landscape in the absence of atypical disturbance of environmental change.

Using ecosystem state at the stand and landscape scales. The following steps can be applied in using ecosystem state as an indicator of biodiversity-based forest degradation.

- Develop or use an existing local forest classification system that reflects available data (see also the ecosystem diversity indicator below). If few data are available, use broad forest type (e.g. open, closed, deciduous, mixed species, moist, and dry). With better data use an ecosystem or forest-type classification (for example, 'mixed-species forest dominated by *Acer* species on mesic soils'), and apply the system over a landscape, comparing actual forest or ecosystem types with predictions based on local knowledge, soil types, known moisture regimes, or known original forest types.

[3] Normal successional stages are not considered to represent degraded states.

- Map forest stands based on their condition using remote sensing or ground surveys and report the area of stands in states other than predicted.
- Report the area of forest that occurs in an unpredicted or undesired state.

Forest fragmentation

Land-use change and other forms of disturbance often lead not only to a reduction in overall forest area but also to the division of remaining forest into increasingly smaller patches, creating new edges between forest and other vegetation types and disconnecting patches from adjacent continuous habitat (Collinge, 1996; Fahrig, 2003; Saura and Carballal, 2004). Usually, a certain amount of fragmentation can occur without significant effects on biodiversity. In some cases it can even lead to higher levels of biodiversity in a given area by increasing the diversity of habitats. Nevertheless, there are system-specific and species-specific fragmentation thresholds that, once surpassed, cause significant biodiversity loss.

Fragmentation has significant and largely negative implications for biodiversity through its impacts on species composition and stand structure; among its effects are a reduction in habitat area, a reduction in 'interior' space (that is, habitat unaffected by edges), increased exposure to edges (where, for example, there may be a greater risk of predation), and spatial and genetic isolation (Fahrig, 2003). Natural ecosystems, especially forests, have become increasingly fragmented on a global scale, which poses a substantial threat to biodiversity (see, for example, reviews by Fahrig, 2003 and Fischer and Lindenmayer, 2007).

Large animals, especially large carnivores, require large areas of habitat and are especially vulnerable to the reduction in habitat area caused by forest fragmentation. Smaller animals can also be affected, and the disappearance of certain species from forest fragments can profoundly affect the forest itself, for example through changes in tree seed dispersal. Even species that persist in fragmented habitat do so in smaller populations, which are more vulnerable to hazards such as disease, predation or the Allee effect (whereby low population densities reduce survival and reproduction rates). Rare species and those that normally occur at low population densities are especially vulnerable to these kinds of effects.

The edges of forest patches are associated with environmental gradients that can affect microclimate, canopy gap formation, biomass and nutrient cycling, regeneration, invasion by alien species, and the risk of predation. For example, invasive alien species are often favoured by an increased incidence of forest edges within a landscape. Fragmentation reduces the movement of species that are reluctant or unable to cross non-forest areas and increases the chance of local extinction of individual species. Overall, these area, edge and isolation effects can singly and in combination adversely affect many forest species and increase their vulnerability to stochastic events, leading to population decline and extinction (Driscoll and Weir, 2005; Arroyo-Rodríguez *et al*., 2007).

Forest fragmentation is a major cause of well-documented reductions in the distribution and abundance of individual species and changes in the species composition of many forest communities, especially in temperate and tropical forests (e.g. Laurance *et al*., 2002; Kupfer, Malanson and Franklin, 2006; Watling and Donnelly, 2006; Ewers, Thorpe and Didham, 2007; Fischer and Lindenmayer, 2007). Beyond certain thresholds, fragmentation may also cause cascading effects on a wide range of ecosystem functions and services (Wu *et al*., 2003; Millennium Ecosystem Assessment, 2005). There is evidence, for example, that forest fragmentation may reduce total carbon storage at the landscape scale (Groeneveld *et al*., 2009) and that hydrological cycles are appreciably altered by forest fragmentation, causing changes in evapotranspiration, local climate and run-off (Ziegler *et al*., 2007).

Fragmentation appears, therefore, to be an excellent indicator of forest degradation for all types of forests, except possibly boreal forests, where, at least in managed landscapes, fragmentation is ephemeral because forests undergo natural or assisted regeneration and other uses (such as agriculture) are generally rare because of climate limitation (Thompson and Welsh, 1993).

Fragmentation needs to be distinguished from normal forest succession following disturbance. Fragmentation is a mixture of forest patches and land that has become deforested and is usually defined as a process involving both the loss and the breaking apart of formerly contiguous habitat. Fahrig (2003) noted that empirical studies of habitat fragmentation are often difficult to interpret because many measures of fragmentation are at the scale of the patch, not the landscape, and most measures do not distinguish between habitat loss (deforestation) and habitat fragmentation *per se* – that is, the breaking apart of habitat after controlling for habitat loss. Fragmentation has come to mean different things to different people and has lumped together many interacting processes and spatial patterns that accompany human landscape modification (Lindenmayer and Fischer, 2007). Figure 4.1 shows some of the processes often described as fragmentation.

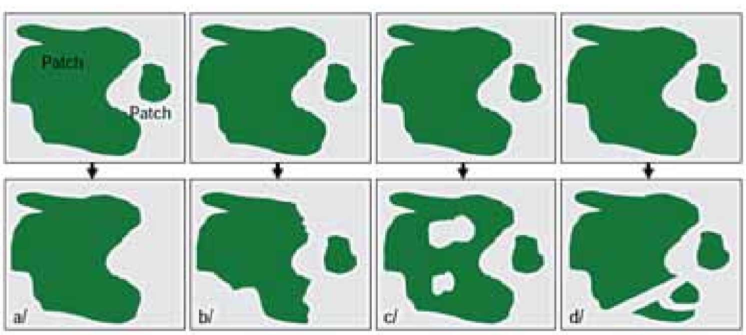
The variation of habitat requirements between species adds further complexity to the use of fragmentation as an indicator. A small arboreal mammal may perceive a road or a cleared area as a barrier – its habitat, therefore, has been fragmented. A large ground-dwelling herbivore, on the other hand, may consider a treeless area as a useful path or grazing resource – its habitat has not been fragmented (and has possibly been improved). Connectivity is species-specific and so, therefore, is habitat fragmentation.

A distinction should also be made between habitat loss and the loss of native vegetation cover because some species can survive or thrive in modified landscapes. Some naturally fragmented landscapes (e.g. the savannah–forest mosaic of coastal Gabon) are extremely species-rich, and a change to 100% forest cover would result in a decrease in the number of species (although it would result in a net gain in carbon stock).

Considering all these elements of complexity, it is suggested that if the baseline condition is a primary forest ecosystem (that may be fragmented naturally) or a sustainably managed

FIGURE 4.1

Four spatial pattern processes of forest loss



a = attrition (patch removed); b = shrinkage; c = perforation; d = fragmentation/breaking apart.
Source: Bogaert et al. (2004).

forest, then an increase in fragmentation over expected natural levels is generally indicative of degradation. There are exceptions: in Costa Rica, for example, an increase in forest area has caused an increase in the fragmentation index because of the establishment of large numbers of small forest patches.

Fragmentation metrics. The most common source of mapped data on forest cover is remote sensing. On the whole, fragmentation data derived from remote sensing at a high spatial resolution, such as from Landsat and *Satellite Pour l'Observation de la Terre* (SPOT), are relatively easier to interpret because they relate directly to forest distribution on the ground. Coarser-resolution remote sensing, such as Moderate Resolution Imaging Spectroradiometer (better known as MODIS), Medium Resolution Imaging Spectroradiometer (MERIS) and SPOT Vegetation, can also be used to assess changes in forest fragmentation but may obscure finer-scale fragmentation important for some components of biodiversity (e.g. species) and some ecosystem services. Data derived from aerial surveys may also be a useful (though expensive) source of forest-cover data for assessing changes in fragmentation. Whichever data are used, it is essential that both the raw data and the ways in which they are processed (including rectification, correction and classification) are comparable for all the time periods being assessed. In some cases this may require specific tests of comparability.

Several metrics can be used to assess fragmentation. Some can be used at the FMU level and others at the patch level. Some fragmentation metrics are more easily understood than others. The most useful potential indicators are those that represent the major effects of fragmentation (i.e. on area, edges and isolation) as transparently as possible.

Table 4.3 shows some possible measures of fragmentation. For all of them, and for most other available fragmentation metrics, their usefulness depends on the level of understanding of their relationship to the values and services of interest. Moreover, in order to minimize confounding effects (as identified in Table 4.3 in the column on caveats and constraints) they must be compatible with and interpretable in the light of information on change in forest area. They are also only useful if measured over time because many forests are naturally patchy or have been fragmented on historical rather than recent time scales. Therefore, establishing a current or recent baseline from which to assess change is essential, as is the use of compatible datasets and analysis methods in consecutive assessments.

Ideally, indicators of forest degradation resulting from fragmentation should be presented in the context of indicators of the pressures leading to that fragmentation (for example, deforestation rates, agricultural conversion and the expansion of infrastructure) and the responses aimed at controlling them (for example, protected-area establishment, other land-use planning and zoning, and the area under certified management). This will help users to interpret trends and to act on their interpretations.

All the metrics listed in Table 4.3 are available in software such as FRAGSTATS (McGarigal *et al.*, 2002) and can be used with data generated using geographic information systems (GIS) or remote sensing at various scales. On their own, however, they do not fully 'measure' fragmentation, and little guidance is available on how they relate to the observed biological (or other) effects of fragmentation (Davidson, 1998). Therefore, they should be complemented by other measurements or assessments to provide a more complete understanding of degradation. It is possible to determine whether fragmentation affects biodiversity in a given forest landscape, but it is less easy to quantify the biodiversity-related

TABLE 4.3

Proposed best fragmentation measures

Metric	Calculation	Most commonly used unit	Relation to degradation	Caveats and constraints
Mean patch size	Total forest area divided by the total number of patches	hectare	Decreasing mean patch size over time is likely to indicate increasing degradation due to area effects	Mean patch size can increase as a result of the elimination of small forest patches
Mean perimeter:area ratio	The mean ratio of the patch perimeter to area for all patches in the landscape	None (dimension-less)	Increasing the mean perimeter:area ratio can indicate increasing degradation, especially via edge effects	The ratio can decline through the elimination of smaller and more complex patch shapes
Mean Euclidean nearest-neighbour distance	The mean distance between all landscape patches, based on shortest edge-to-edge distance	metre	Increasing mean nearest neighbour distances are likely to indicate increasing degradation due to the effects of isolation	The loss of individual isolated patches can cause a decrease in the mean nearest-neighbour distance
Forest integrity index (e.g. Kapos, Lysenko and Lesslie, 2000)	Combined metrics of patch size, connectivity and edge effects	None (dimension-less)	A declining integrity index is likely to indicate a reduced ability to produce goods and services and therefore increasing degradation	The relationship to specific goods and services not established – complexity may obscure more understandable trends

degradation of a landscape due to fragmentation, or to apply results from one landscape in others. For example, point data can be used to assess species' responses to habitat edges (Ewers and Didham, 2008), but to quantify the landscape-scale net impact on the populations of those species it is necessary to combine information on species' responses with spatially explicit data on the distribution of habitat edges (Ewers and Didham, 2007; Ewers, Thorpe and Didham, 2007; Ewers *et al*., 2009).

Calculating fragmentation indices. The calculation of fragmentation indices involves the following:

- The use of GIS and software such as FRAGSTATS to analyse data generated by digitized aerial photographs or high-resolution satellite imagery.
- The comparison of historical and current data, or the comparison of current data and predictions of natural fragmentation patterns.

Ecosystem diversity

An ecosystem can be defined as a dynamic complex of plant, animal and micro-organism communities and their non-living environment. Classifications of ecosystems can be made at any scale, from global classifications, such as sub-biomes, to local ecological communities, such as the classification of forest stands based on vegetation associations and a characteristic set of tree species (e.g. Allen and Hoekstra, 1992). In many countries, classifications of forest vegetation types are also used as classifications of ecosystems. The use of ecosystem diversity as an indicator requires knowledge of, or an ability to predict, the pre-existing distribution of ecosystems in a landscape.

The ecosystem diversity indicator suggests an expectation that, within bounds, a certain percentage of a landscape should be in each of several known forest types, and that the broad species composition (e.g. multiple species, conifer or deciduous) of a forest stand should be predictable given certain pre-existing conditions. Each ecosystem has a characteristic biodiversity that is recognizable. Since biodiversity supports almost all ecosystem goods and services, the loss or degradation of biodiversity in any ecosystem type, especially the loss of functional species, will result in a reduction of ecosystem goods and services (e.g., Diaz *et al.*, 2005). Therefore, the use of ecosystem diversity as an indicator is useful for suggesting broad changes in the range of forest values that are produced across a landscape.

At a minimum, the forest types used by the United Nations Environment Programme-World Conservation Monitoring Centre (UNEP-WCMC, 2007) or FAO's ecological zones (FAO, 2001) may be used to the level of sub-biome to classify ecosystem diversity. Measurement of the extent of ecosystems or habitats is usually made using remote sensing techniques such as aerial photography or satellite images, and GIS for analysis. Time series using identical classifications of data enable the analysis of change over time in a sampled area.

Landsat, ASTER, SPOT High Resolution Visual, and Indian Remote Sensing satellite imagery, with spatial resolutions of 15–60 m, have been used for forest mapping at the national and subnational levels (Strand *et al.*, 2007). However, maps generated using imagery at this scale provide only rough estimates of forest type and structure; often there is even difficulty in distinguishing plantations from natural forests (Strand *et al*., 2007). For example, UNEP-WCMC (2007) mapped four classes of plantation (temperate/boreal exotic species plantation, temperate/boreal native species plantation, tropical exotic plantation, and tropical native plantation) at a coarse scale using Advanced Very High Resolution Radiometer-based satellite images, but the dataset was too coarse (20 m) to be useful for the more nuanced identification of forest ecosystems.

Souza *et al.* (2003) developed a method for mapping degraded forest classes, which they defined as heavily burned or heavily logged and burned, using a combination of 1 m resolution IKONOS data and data from SPOT 4. Even at this resolution, however, tree species could not be identified with accuracy, meaning that it could only be used for broad forest typing (e.g. deciduous, conifer, open or closed). Lambin (1999) noted that images must be evaluated at sufficient frequency to differentiate natural forest change from degradation.

Thus, satellite imagery can be used to assess change in ecosystem diversity at a scale of broad forest types (or ecosystems), but not for finer-scale forest types. The techniques require expertise that may not be available in all countries and a forest classification system against which to measure change. Fine-scale assessments require expensive imagery (e.g. LiDAR – 'light detection and ranging') and highly specialized study. Mid-resolution remote sensing can be used to generate a first approximation of change in the relative abundance of ecosystems, such as the relative abundance of dry and wet tropical forests or of conifer and mixed-species temperate forests.

The indicator to be measured is 'change in area/percent of forest ecosystems', against a baseline condition. This could be reported using any one of a number of similarity indices that compare locations or the same location over time. Most simply, Sorensen's index of similarity could be used, in this case to measure the difference between actual and expected landscape structure, as expressed in the following equation.

$$\textit{Sorensen's index of similarity} = \frac{2z}{x + y}$$

Where
x = the number of forest types in the landscape of interest
y = the number of forest types in the reference landscape or at time t+1
z = the number of ecosystems common to both.
The index takes a value between 0 and 1, where 1 = no difference.

For multiple landscapes, Whittaker's formula, as modified by Lennon et al. (2001) or Diserud and Odegaard (2007), could also be used:

$$b = \frac{2|b - c|}{2a + b + c}$$

Where
a = continuity (i.e. the number of same forest types in both landscapes)
b and c are exclusive forest types and c co-occurs on the landscapes or on the same landscape at different measurement times. (The symbol around b - c indicates absolute value, e.g. |4 - 6| = 2.)

Monitoring landscape-scale ecosystem diversity. The monitoring of landscape-scale ecosystem diversity involves the following steps:

- Use FAO's forest ecosystems as a first approximation, or a national or regional forest-type classification system, to measure the relative abundance of forest types.
- Map an area using the best available imagery, with ground-truthing if possible, for selected classes (ecosystems). The area to be mapped could be a large production landscape, an FMU, or a sufficiently large area (e.g. 100 km^2) across which to sample forest ecosystems.
- Develop an *a priori* expectation of forest types for a given landscape based on the natural range of variability (NRV) for those forest types obtained from historical information or from a nearby primary forest landscape on similar site types.
- Monitor change in ecosystems (area, percent) in the area of interest, at a time interval that is appropriate relative to natural and anthropogenic disturbances.
- Use the NRV to bound the occurrence (area, percent) of each forest type as a means for determining when degradation is occurring as a result of human activity.
- Calculate an index of similarity.

Monitoring stand-scale ecosystem diversity. This scale could also be referred to as habitat diversity or forest types, as it has been by the Biodiversity Indicators Partnership.[4] The most widely used techniques for stand-level remote assessments use small-scale aerial photography (e.g. 1:20 000). High-resolution satellite images can be used at the scale of the individual tree but are generally cost-prohibitive. Recently, LiDAR and other aircraft-mounted sensors have become more common in small-scale forest mapping, but they are expensive. Most work at the stand level to assess degradation will likely involve ground surveys to sample change in forest condition.

The degradation of ecosystems at the stand scale may be best assessed using other indicators, such as biomass production, species occurrence and bushmeat production.

[4] http://www.bipindicators.net/.

Forest tree species

Achieving a similar (although perhaps skewed) composition of forest tree species in managed compared with unmanaged stands is an underlying tenet of SFM.[5] A significant (unplanned) departure from the expected species composition can suggest degradation of the goods and services produced by an ecosystem and may indicate a loss of biodiversity with respect to tree species and other organisms such as lichens, fungi and insects. Tree species composition can change as a forest becomes degraded through the over-harvesting of commercially valuable species, excessive fire, or other unsustainable practices (for examples see Asner, Knapp and Broadbent, 2005 and Foley *et al*., 2007). The tree species composition of a stand provides fundamental information on changes in tree community diversity, from the stand level to the regional level, and ultimately on forest ecosystem stability over time. Using tree species composition as an indicator requires an understanding of forest ecosystem types (see above), their 'normal' tree species compositions, and the variance found in similar ecosystem types across landscapes (known as beta diversity). Community-level analysis also requires knowledge of successional processes for given forest types.

It may be difficult to establish a baseline, especially for humid tropical forest types, where differences in diversity between plots are high because of the high diversity of tree species at low densities. Moreover, the use of tree species composition in a primary forest to define baseline composition raises questions such as the amount of difference between the primary forest and the targeted forest that is acceptable, which primary forest (if there are several) should be compared with the target forest, and how the baseline composition can be established if no information is available on a relevant primary forest. Since species in similar vegetation tend to occur in clusters (Yoshimura, 2007), tree species composition in degraded forest might need to be defined locally or at the landscape level. Highly degraded forests will show large differences, to the extent that the state of the system may have changed. Hence, surveys of tree species composition can be used to supplement remote sensing that indicates that the forest is in a new ecosystem state (see 'ecosystem state' indicator above).

Generally, information on expected species occurrence is available from local-scale to national forest inventories, especially where adequate forest management planning processes are in place. A range of analyses is available to determine meaningful changes in tree species composition by comparing community structure, including discriminant function, clustering, and the use of various simple indices (e.g. those suggested above for ecosystem diversity).

Determining tree species composition. Numerous plot-based and plotless techniques can be used for tree species composition surveys (e.g. prism plots, single large plots, multiple smaller plots, point-quarter, and point-distance). The technique used to census species is probably less important than an adequate design for the monitoring programme and maintaining consistency with past surveys (if they exist). The following steps should be taken to prepare for a tree species census:

- Use the FAO ecological zones as a first approximation, or a national or regional forest type classification system, to determine the relative abundance of forest types.
- Select forest types of interest for surveying based on relative abundance (most common and most heavily used) and regional priorities (e.g. rare forest types).

[5] The species mix need not be identical to primary forests, however. Enrichment planting or other forms of silviculture may be used to increase the abundance of selected species valued for their products (wood or non-wood). In the teak forests of Southeast Asia, and certain forests in the Amazon and Central America, for example, forest-dependent people have significantly altered species composition near settlements for their own benefit.

- Map an area using the best available imagery, with ground-truthing if possible, for selected classes (ecosystems). The area to be mapped could be a large production landscape, an FMU, or a sufficiently large area (e.g. 100 km^2) across which to sample forest ecosystems.
- Develop an *a priori* expectation of species abundances in each forest type for a given landscape, based on the NRV for forest types derived from historical information or a nearby primary forest in a similar landscape.
- Determine the number of plots to sample based on the expected variance among plots.
- Conduct a field study using plot-based or plotless methods. Once a method is selected it should not be changed. Equipment will include maps, data loggers or field notebooks, prisms and measuring tapes.
- Determine means and standard deviations for each forest type and develop simple indices of similarity (e.g. Sorenson's), or use appropriate multivariate ordinations to examine for differences between sampled stand types and controls. On a multivariate plot, stands that are similar to controls (or expected values) will cluster near values for the controls, while disparate stands will tend to be elsewhere on the ordination.

Functional species

Any change in forest type (e.g. age, vegetation or structure) and abiotic environment in and around forests (e.g. average, highest and lowest temperatures, precipitation or snow accumulation) will result in a change in associated species composition. A subset of species composition of particular interest is functional species (e.g. Diaz and Cabido, 2001; Ellison *et al.*, 2005). Some species are more important than others in providing ecosystem goods or services (e.g. Walker, 1992; Diaz *et al.*, 2003), and the loss of functional species often means a reduction in a given function and the reduced production of goods or ecosystem services. There is often redundancy among species for a given functional role, however, and functional groups (i.e. groups of species that perform the same ecosystem function) can also be used as indicators of forest degradation. The loss of functional species in the absence of redundancy has negative consequences for ecosystems to the point of ecosystem change or even collapse (Chapin *et al.*, 1997; Ellison *et al.*, 2005). Hence, the loss of functional species often means a reduction in a given function and in the production of goods or ecosystem services. Expanding the concept to include redundancy, a change in functional groups (groups of species that perform the same ecosystem function) can be a strong indicator of ecosystem change. For example, the loss of all pollinators would have highly negative consequences for plant reproduction. Keystone species are a special group of functional species that carry out roles in ecosystems that affect many other species; where they occur, they are indicators of ecosystem functioning and, if lost, indicate forest degradation. Functional species and keystone species are not necessarily the most numerous species in the system (see Hooper and Vitousek, 1997; Diaz *et al.*, 2003). Table 4.4 lists possible species indicators of degradation in ecosystem function.

If functional species in ecosystems can readily be identified, changes in their abundance over time may indicate that the system is being degraded. Below, several ideas for monitoring functional species as indicators of degradation are suggested. The use of any of these requires a monitoring programme that should be carefully designed locally by people working at the stand level. Hence their applicability as tools to assess degradation can only be accomplished by local managers, given sufficient training and planning.

Many species deliver regulating services in ecosystems. For example, certain birds, butterflies and ground beetles are often used as indicator groups because of data richness and because many are clearly functional species (Lawton *et al.*, 1998) that deliver services such as pest reduction and pollination. Insectivorous birds can regulate herbivore populations and act as seed dispersal agents, and some are pollinators. In conifer forests in the western United States, bird predation on chronic insect herbivores was shown to increase productivity in forest stands by as much as 20 percent compared with control sites with no predation (Bridgeland *et al*., 2010). Most insectivorous birds respond negatively to selective logging and partial harvesting in tropical forests (e.g. Johns, 1996; Mason, 1996; Aleixo, 1999), suggesting that this guild may be a good indicator of degradation. Ecological information is also richer for insectivorous birds than for many other functional species. For example, when species or age composition becomes skewed, such as when only early-succession-forest birds are recorded at the landscape level, it could suggest degradation at that scale (Yamaura *et al.*, 2009). On the other hand, ground beetles are more indicative of stand condition than landscape condition. Monitoring methods are well established for both species groups (e.g. point counts for birds and pit-fall traps for ground beetles). According to Whittaker (1972), gamma diversity is the richness in species of a range of habitats in a geographic area (e.g. a landscape or an island); it is dependent on the alpha diversity of the individual communities and the range of differentiation (beta diversity) among them. Like alpha diversity, gamma diversity is a quality which simply has magnitude, not direction, and can be represented by a single number (a scalar).

If species information, which was collected originally at the stand level, indicates higher species diversity in natural forests than in plantations at the landscape-to-regional level, where there are patches of each forest type, then species composition might be a good indicator of degradation at that scale. The difference in species composition at the landscape level, therefore, might be a useful way to represent forest degradation.

Pollinator bees as indicators. Pollinators are directly or indirectly related to ecosystem productivity (e.g. Ricketts *et al.*, 2004; Klein *et al*., 2007). The importance of the various pollinator groups differs by ecosystem or ecoregion. For example, insect pollinators are most important globally but bat pollinators occur mostly in the tropics, while birds (e.g. hummingbirds, honeyeaters and sunbirds), mammals and sometimes lizards can play important pollination roles in particular regions or ecosystems.

In Southeast and East Asia, bees in the tribe Apini of Apidae, including *Micrapis*, *Magapis* and *Apis*, are particularly useful indicators of forest degradation because they are important pollinators of tree flowers and they nest in forests. Thus, a dominant bee species may be a keystone species. Many hover flies are also important pollinators that inhabit forests. They are often associated strongly with woody debris, but their ecology is generally not well known.

The population size of key pollinator species can be an indicator of ecosystem function and degradation. The ratio of plant species that require biotic pollination to those that do not might indicate the condition of pollinators in a forest ecosystem. As most key pollinators (especially bees) require relatively large dead standing trees in which to nest, the presence of these can be used as an indicator of the presence of bees.

Bee fauna can be surveyed as follows (see Westphal *et al.*, 2008):

- Select several stands (e.g. five or more per forest type) for sampling to enable a measure of variance among stands.
- Trap bees with pan-traps (yellow, white and blue are the standard colours), which are effective for landscape-scale sampling. For some species, trap nests with reed/bamboo internodes can be used as a complementary sampling method.
- Identify trapped species, preferably at the morpho-species level but at least at the family level. Count the number of individuals of each family.
- Determine a baseline for bee species using one or more of the following – total species numbers, number of indicator bee species, composition of specialist vs generalist, or bee

diversity index. Communities can be characterized and compared using diversity indices such as

- calculating species richness for stands and forests (i.e. the number of species)..
- the Simpson index (often used to measure the diversity of habitats), which can be used to describe bee diversity, as follows

$$D = 1 - \sum_{i=1}^{S} p_i^2$$

Where D is the diversity index, S is the number of species in the sampled community (stand), and p is the proportion of the ith species in the sample. D in sampled stands can be compared to D in controls.

- the Shannon index (denoted by the letter H), which can be used to indicate species diversity as follows

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

Where S is the number of species in the community sampled (e.g., in the stand) and p_i is the proportion of S made up by the ith species. In this case, H in sampled stands can be compared to H in the control stand.

Regardless of which index is used, it is important to compare the relative abundances of the individual species to allow direct comparisons between samples and control stands (or expected values).

Seed-dispersal agents as indicators. Seed dispersal is related directly to natural forest regeneration and therefore to ecosystem sustainability. The seeds of many tree species are dispersed by animals, while others are dispersed by abiotic agents (e.g. wind and water). Many plants (known as zoochoric plants) produce fruits whose seeds are intended to be carried by vertebrates. Therefore, those plants are important food resources for animals. For many vertebrates, zoochoric plant richness and their annual rate of reproduction (the amount of fruit set) is highly related to reproductive success. Birds are generally one of the most important groups of seed-dispersal animals in forest ecosystems, and primates are also important seed-dispersers in tropical forests (e.g. Chapman, 1989). Rodents can carry nuts but are also nut predators (Howe and Smallwood, 1982).

Many key seed-dispersers (e.g. birds, bats and primates) move relatively long distances within a given forest, and the loss of their habitats at the landscape scale might limit or prevent the natural regeneration of some trees (Gorchov *et al.*, 1993). Moreover, some small plants, such as *Viola* species, also rely on insects as seed-dispersers. As shown in the *Viola*–ant system (Hanzawa, Beattie and Culver, 1988), germination may also be highly dependent on the seed-disperser system.

Habitat conditions (e.g. the area of natural forests, the extent of fragmentation, the connectivity of forest patches, the existence of corridors, and the density of dead standing trees and relatively large living trees as nests) can be used to predict the presence and/or density of certain seed-disperser animals. The abundance of zoochoric plants and their rate of regeneration would likely be a predictor of the condition of populations of seed-dispersers.

TABLE 4.4

Possible species indicators of degradation in ecosystem function

Indicator	Potential as indicator	Measurement method	Relevant case studies	Measurement scale
Pollination				
Population size of key pollinator species	+	National inventory		Landscape
Habitat quality for key pollinator species	+	Number of snags, old (and maybe large) trees		Stand, landscape
Natural vegetation (in some senses, this may be the same as habitat quality)	+	Area of natural vegetation; distance between patches of natural vegetation	Gathmann and Tscharntke (2002); Taki *et al*. (2010)	Landscape
Seed dispersal				
Habitat quality for key seed dispersers	+	Area of natural forest; fragmentation; connectivity of each forest/corridor; snags and relatively large trees as nests		Stand, landscape
Zoochoric plants (for seed dispersers)	?	Number; amount of young zoochoric plants		Stand, landscape
Seed dispersal animals	+	Species richness; population size; national survey		Stand, landscape
Decomposition				
Soil animals for decomposition	-	Species composition	Mostly used in scientific study, requires expertise (Ritz *et al*., 2009)	Stand
Soil physical and chemical properties	+	National and local survey		Stand
Biological control				
Natural enemies for biological control	?	National and local survey, species richness		Stand, landscape
Insectivorous birds and bats	+	Species richness; species composition; population size		Stand, landscape
Habitat quality for biological control agents	+	Natural vegetation; forest connectivity (fragmentation)		Stand, landscape
Period of outbreak of pests (for evaluation, need more scientific evidence)	?	National survey		Stand, landscape
Carbon sequestration				
Relevant major tree species for carbon sink	?	National survey, inventory	Russell *et al.* (2010)	Stand
Tree growth	+	National survey, stand-level survey		Stand
Soil nutrition level	?	National survey	Oren *et al*. (2001)	Stand

Note: - = limited potential as indicator; + = significant potential as indicator

Decomposers as indicators. Decomposers in forests help to maintain water and soil quality and promote nutrient cycling (Harris, 2009). Microorganisms are probably the most important forest decomposers, but little qualitative and quantitative information is available on them or how they function (e.g. Meyer, 1994; Harris, 2003). Microorganisms decompose organic materials from the macro-scale to the micro-scale throughout the decomposition process. The process of decomposition in forests occurs in stages, with many organisms dependent on products from preceding stages. However, the specific organisms associated with the various stages of decomposition are mostly unknown.

The soil microbial community is dependent on the level of site disturbance. The nature of the soil microbial community, therefore, can indicate the impact (success) of restoration and management practices (Harris, 2003).

At the micro-scale it has been shown that different wood-boring insects and wood-decaying fungi prefer different dead (or almost-dead) trees. Soil organisms are generally more species-rich in litter with higher nutrient loads (Hättenschwiler, Tiunov and Scheu, 2005). Therefore, at least for soil formation in the early stages of organic decomposition and nutrient cycling, soil organism diversity is crucial for maximizing these soil services (Harris, 2009).

The most important groups of decomposers are micro-organisms, but there is no 'best' technique for monitoring them, and all techniques require specific expertise and equipment (Harris, 2003; Ritz *et al.*, 2009). The species composition of soil animals is sometimes used as an indicator (Yeats, 2003). However, there are difficulties in the identification of soil animals and therefore they are usually only used in scientific investigations rather than for operational monitoring. DNA markers can be used, but only in small-scale studies. Nevertheless, the use of such techniques can help to differentiate the stages of soil degradation and to identify functional relationships with above-ground primary production. The suite of methods available for developing a monitoring programme for decomposers is described in Ritz *et al*. (2009).

Biological control species as indicators. Insect herbivores are common in forests. Biological control agents include pathogenic microorganisms, insect predators and parasitoids (Debach and Rosen, 1991; Maleque *et al*., 2010), and insectivorous birds and bats (Kalka, Kunz and Fenton, 2008). The importance of each group may differ locally, as does species composition.

Generally, there are relatively few insect predators in natural forests; when pest species increase in abundance, their natural predators respond by increasing numerically as well, but with a time lag. Thus, in natural forests, increases in the abundance of herbivores often do not last long because they may be controlled by natural predators, although population 'escape' by a pest species is possible, resulting in considerable damage to forests (e.g. Logan and Powell, 2001).

Some predacious insects, such as ground beetles and parasitoid wasps, do not migrate long distances and forest fragmentation may limit their distribution (Kagawa and Maeto, 2009). Birds and bats show particular habitat preferences. Although some ground-dwelling beetles are sensitive to habitat conditions, including vegetation (e.g. Nummelin and Fursch, 1992), ground ants may not be highly influenced by changes in vegetation cover (e.g. Oliver, Nally and York, 2000; Vasconcelos, Vilhena and Caliri, 2000).

The period of a pest outbreak is one indicator of the strength of biological control. For example, an outbreak of invasive alien species tends to continue until the prey or host populations have collapsed because they have no natural predators in the invaded region.

Important predaceous insect groups are ground-dwellers, and pit-fall traps are therefore useful methods for sampling them. For ants, bait traps can be used but line transects to count colonies are an alternative for well-trained technicians. Bamboo/reed trap nests work for some wasps; such nests also provide information on their prey diet. A malaise trap is generally used for collecting flying insects. Usually a tally is made of the number of individuals of each species trapped per day.

Identification is relatively easy for ground-dwelling predators. In all cases, managers need to establish a sampling protocol using control and managed stands and to develop a trend index of the abundance of species or groups of species over time against the control stand to determine whether forest degradation is occurring.

Species important as forest carbon sinks. Tree growth rates and increases in biomass are indicators of change in the above-ground carbon sink. Not all trees store carbon equally (e.g. Strassburg *et al.*, 2010; Russell *et al.*, 2010; Potvin *et al.*, 2011; but see Zhang *et al.*, 2011), and species may be short-lived or long-lived. Although there is only an incomplete understanding of the relationship between carbon sequestration and tree species and between carbon sequestration and forest management and disturbance, some tree species require more carbon than others during growth and therefore store more carbon (e.g. Russell *et al.*, 2010). As a result, forests may be degraded in terms of the carbon storage services that they supply if the tree diversity is reduced. Hence, the monitoring of species richness and especially the individual monitoring of carbon-dense tree species density over time, or in comparison to an expected tree density, can indicate degradation of a forest ecosystem (see ‘stand-scale ecosystem diversity’ above and ‘biomass indicators’, also above).

Provisioning species. Indicator species should be determined locally and nationally by considering the relative importance of each species for various provisioning functions (e.g. timber, food and medicine). Certain plants may be culturally important for NWFPs. Forest degradation can be measured by yield as the difference between expected and actual yields. These yields are related to the stand-level biodiversity of the forest ecosystem and the loss of yield is therefore related to the degradation of the biodiversity.

Invasive alien species

In forests, an invasive alien species is a species not native to a given forest type that has invaded the forest and is causing harm (Pimental, Zuniga and Morrison, 2005). The invasion of alien species can cause a change in forest state and a consequent reduction in biodiversity and other forest goods and services. Many forests have been degraded by invasive alien species (e.g. Chornesky *et al*., 2005) and most indicator processes therefore make use of this phenomenon as an indicator. It is included in this chapter because the usual effect of invasive alien species is to reduce native species, through either competition, herbivory or predation (e.g. Lucier *et al.*, 2009).

In some circumstances, the spread and impacts of specific invasive tree species can be mapped using remote sensing (Van der Meer *et al.*, 2002). For example, certain invasive alien species occur in or dominate forest canopies and have been mapped remotely; such species include tamarisk (*Tamarix chinensis*) (Everitt and Deloach, 1990), leucaena (*Leucaena leucocephala*) (Tsai, Lin and Wang, 2005), maritime pine (*Pinus pinaster*) (Ferreira, Aguiar and Nogueira, 2005), Chinese tallow (*Sapium sebiferum*) (Ramsey *et al*., 2002), and Australian wattles (*Acacia* species) (Theron *et al.*, 2004). Another valuable use of remote sensing in monitoring invasive alien species is the effect that some invasives have on forest condition. In Hawaiian montane rain forest dominated by *Metrosideros polymorpha*, Asner and Vitousek (2005) used aircraft with an infrared imaging spectrometer to show that leaf nitrogen concentrations were reduced in forests invaded by *Myrica faya*. In some cases, remote sensing can detect invasive tree species where they have differential morphology or coloration. For example, Pauchard and Maheu-Giroux (2007) used the yellow colour of *Acacia dealbata* to estimate the extent of its invasion of forests in Chile using 1:20 000 digital colour aerial photographs on a 30 x 30 m grid. Limitations on these kinds of data include the availability of suitable technology, the expert capacity to analyse the data, and the cost of acquiring the imagery.

Invasive insect herbivores, such as emerald ash borer (*Agrilus planipennis*), and pathogens, such as Dutch elm disease (*Ophiostoma ulmi* and *O. novo-ulmi*), have caused the degradation of millions of hectares of forests in North America. In many cases, damage to forests can be mapped by counting the number of dead trees using aerial photographs. Damage caused by defoliating insects can also be mapped remotely if it is severe enough to be detected by the sensors.

Other invasive insects, such as ants and earthworms, can cause cascading effects (i.e. effects that permeate through the system and result, for example, in the reorganization of community structures) over large areas as a result of competition or the replacement of endemic species in systems (Kenis *et al*., 2009; Straube *et al*., 2009). However, changes caused by these kinds of species are often subtle and difficult to monitor.

Even if sufficient funding is available for a large-scale remote sensing study of an invasive alien species as an agent of forest degradation, measurement at two or more points in time is required. Images will need to be acquired at intervals of several years to detect trends and to provide a measure of change over time. Nevertheless, the current extent of an invasive alien species can be determined using a single set of images.

Landscape-scale and stand-scale monitoring of invasive alien species. An indicator of forest degradation is the area of forest damaged by invasive alien species. The actual method will vary depending on the technology and expertise available, the nature of the invasive alien species and the type of damage it causes. In some cases modelling can be used to predict the area affected.

Invasive alien species can be monitored as follows:

- Develop a list of invasive alien species and map their distributions using remote sensing if possible, ground surveys and/or *ad hoc* observations.
- Assess the impact of each invasive alien species using expert opinion and available research.
- Monitor change in the distribution of invasive alien species over time, and their impacts on biodiversity.

MEASUREMENT FREQUENCY AND REPORTING

The abundance of most animal species used as biodiversity indicators will vary naturally from year to year, so the more frequently that a species (or community) is monitored, the more rapidly trends over time can be detected. In the case of tree species diversity, ecosystem diversity and fragmentation, change is much slower and these indicators need only be monitoring every few years to observe a trend or to provide an indication of degradation against a known or expected baseline value. Fragmentation, for example, tends to be a continuous process and so managers must assess change necessarily over time and against an expected landscape condition.

ISSUES AND CHALLENGES

The determination of the range of natural variation for many biodiversity indicators is a major challenge. The task is easier, however, if unmanaged forest is available for use as a control than in situations where data from other areas must be used to set a baseline. Forest managers and stakeholders will have to decide the level of indication of degradation that they are willing to accept. For example, ecosystem state and fragmentation should constitute the minimum level of reporting necessary. The use of animal and plant species or community indicators requires considerably more effort, usually at ground level, but they indicate a much more insidious and important level of degradation than can be broadly determined through remote sensing and could provide evidence that degradation has occurred long before there is a change in state or fragmentation causes a noticeable effect. The challenge for managers is to develop a programme and maintain it over time to determine whether management is degrading the forest at broad and local scales.

A related challenge is the issue of the wide variance in species abundances across years and the need for extensive sampling to obtain statistically useful results (e.g. Rolstad *et al.*, 2002) and to be clear that the selected species indeed indicate habitat (e.g. Niemi *et al.*, 1997). Indicator species analysis is a key way forward in the use of indicators as analytical tools (e.g. Bakker, 2008). In all cases, indicators should be selected carefully and with the best expert judgement possible considering feasibility and relevance to the issue. Good judgment remains the foundation of competent indicator selection. Data and expertise can inform this judgment. Importance should also be placed on improving skills and institutions, as these are ultimately the building blocks of effective forest management (Shiel, Nasi and Johnson, 2004).

5. Production of forest goods

Forest goods include timber, NWFPs and fuelwood (including charcoal). Forests also provide a wide range of ecosystem services, but these are not considered here for reasons outlined in the introduction. This chapter focuses particularly on natural tropical forests, where a large part of today's forest degradation is taking place. It should be read in conjunction with Chapter 3, which sets out methodologies for monitoring forest growing stock and biomass, and Chapter 4 on biodiversity because forest goods are derived from biodiversity.

Temporary reductions in the production of forest goods may be a part of SFM or the result of natural causes, but a long-term reduction is a form of forest degradation. A reduction in the availability of various socio-economic benefits from forest goods can be assessed through surveys of their availability and use among local populations.

The long-term reduction in the supply of forest goods as a result of over-harvesting is addressed here. In practice, this is often assessed by comparing the removal of forest goods with a level judged to be sustainable, as typically defined in forest management or wildlife management plans.

Stakeholders may mistakenly view short-term changes in production as degradation. This arises from the misperception that a forest stand is the basic unit of decision-making for the conservation or enhancement of productive capacity and the provision of forest ecosystem services. In general, however, forest management decisions are based on planning across a territorially designated FMU, which may be a holding, a forest estate or another type of land unit (e.g. a watershed). Depending on their size, these FMUs typically consist of at least dozens of stands or management compartments with differing species compositions, tree ages and other structural characteristics. The mix of individual stands in an FMU is subject to constant change due to biological processes and management interventions. For example, there may be reductions of carbon stock in some stands over the course of a year as a result of harvesting or burning, and increases in other stands as a result of biological growth. It is the scale of this territorial entity – the FMU – at which management objectives should be set and changes in forest production should be managed and assessed.

Over-harvesting in an FMU can be for timber, NWFPs, fuelwood, bushmeat or fodder. The underlying cause of over-harvesting is often the short-term economic benefit that can be gained from the sale of the harvested goods, but it can also be a greater-than-sustainable demand by people meeting their subsistence needs. Typical contributing factors to over-harvesting include a lack of proper planning and control of forest use, a prevalence of illicit activities, weak governance, a lack of awareness and knowledge on how forests should be sustainably managed, and unclear tenure.[6] The wide variation in local forest situations and management objectives should be considered when defining degradation indicators on the basis of the supply of forest goods.

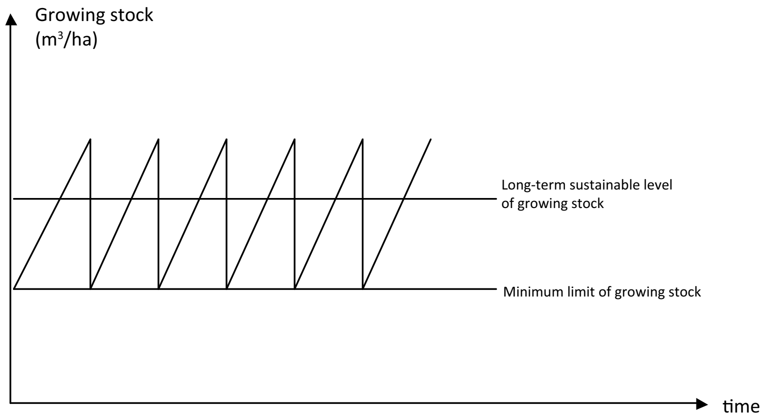
When primary natural forest is subject to (persistent or intense) human intervention, the outcome is typically a change in biomass and species composition. Modified forests[7] can be managed sustainably for production while maintaining other values, even though they

[6] Unclear or disputed tenure can lead to a 'tragedy of the commons', whereby the actions of many people acting in their own short-term self-interest ultimately deplete a shared resource.

[7] The term 'secondary forest' is sometimes applied to modified natural tropical forest which has been subject to selective logging (e.g. FAO, 2006).

FIGURE 5.1

Schematic illustration at the stand/compartment level of periodic change in growing stock due to harvesting (selective logging), tropical forest



Note: Degradation occurs if harvesting reduces the growing stock below a minimum limit. In many tropical forests, the regulation of harvesting level is based on a minimum allowable dbh (i.e. trees with dbh greater than the minimum may be harvested).

differ from the primary forest. Modified forest, therefore, cannot be considered to be degraded *a priori* on the basis of indicators related to production (although it may be degraded on the basis of other indicators, such as those related to biodiversity).

The degradation of the provision of forest goods should be assessed at well-defined temporal and spatial scales. In general, assessments should be made over a sufficiently long time to enable the effects of natural fluctuations and forest management interventions to be taken into account. Figure 5.1 shows how the average volume of biomass per hectare can vary in the short term due to the selective harvesting of large commercial trees.

WHAT AND HOW TO MEASURE

Table 5.1 provides a list of indicators of degradation in the provision of forest goods (timber, NWFPs and fuelwood). It also provides a list of possible sources of the information required to put such indicators into use and outlines the limitations of each potential indicator.

In forest management for timber, it is useful to set thresholds for ‘degraded’ and ‘non-degraded’ forest on the basis of indicators such as mean annual increment (MAI), canopy cover or stocking density. Indicators of over-harvesting include reduced yields, reduced or altered population densities of certain species, diminished reproductive capacity both at the (commercial) species and targeted population levels (as evidenced, for example, by abandoned or exhausted coppices in certain forest types), and deterioration of soil quality (due to harvesting damage or inappropriate forest road construction). In addition to reducing the carbon stored in aboveground biomass, forest degradation may also reduce future biomass accumulation by causing a shift in species composition and tree size structure from mature forest trees to pioneer tree species and woody vines (lianas).

In the case of the informal or subsistence use of forest goods, evidence of forest degradation can be assessed by monitoring sequential reductions in the abundance, density and/or size-class distribution in a forest over time or over a given distance from a defined point (Figure 5.2 presents a useful model by Ahrends *et al.*, 2010). It is also possible to use social and socio-economic indicators as proxies for degradation, with the caveat that these variables will need to

TABLE 5.1

Potential indicators of degradation of the provision of forest goods and services

Forest good	Potential indicators of degradation	Potential for up-scaling	Potential sources of information	Limitations
Timber	Shortened cutting cycles AAC exceeded Canopy cover Number of harvested trees below the established minimum diameter Absence/inadequate number of commercial-sized specimens in logged forest Inadequate number of juvenile specimens of selected species Absence/inadequate number of designated seed trees for a given species in a given logging compartment Reduced supply of timber species in regional markets	Medium to high – national reporting (e.g. FAO, ITTO) on annual timber production can be used against independently gathered data	Forest inventories at national, subnational and FMU levels Pre and post-harvest inventories Forest cover maps Expert assessments Permanent sample plot data in logging concessions Herbarium data Inspection reports of enforcement authorities Audit reports from certified forests Sawmill input statistics	Absence of long-term data Absence of reliable national data Inadequate enforcement activity Confidentiality of certification audit reports Inconsistencies between datasets
NWFPs	Changes in plant population structure over time or space and replenishment of selected species Decrease in yield/locally gathered volume Reduction in recorded production, consumption and export Negative changes in revenue from NWFPs, average time taken or average distance travelled to collect a specified volume of certain NWFPs, or their importance in household resource portfolios (indirect measurement) Increasing prices Reduced market supply	Low to medium	Largely locally collected data through survey and other methods National data on collected/ consumed/traded volumes and NWFP income Foreign trade data for export data Periodic assessments of NWFP population structures Comparison of permit data for allowable harvest with local supply and demand surveys and other data Expert assessments	Largely locally based and intensive; participatory approaches are almost always necessary Overall lack of national statistics on NWFPs, especially in developing countries Lack of enforcement records Lack of NWFP-species management systems
Fuelwood	Reduction in the supply of fuelwood and charcoal Reduction in subsistence consumption of fuelwood and charcoal Negative changes in revenue from fuelwood, average time taken or average distance travelled to collect fuelwood, or the importance of fuelwood in household resource portfolios	Low to medium	Inventory data on above-ground biomass from sample plots and above-ground biomass increment data to estimate potential supply Local and aggregated production and consumption data, including from surveys among users and producers Data from bioenergy plantations Comparison of permit data for allowable harvest with local supply and demand surveys and other data	Exclusion of dead wood and fuelwood from non-forest areas (trees outside forest) can distort supply estimates

be adjusted for other factors that may influence change. Social indicators can help in validating information obtained through direct measurement. For example, Kleine, Shaharuddin and Kant (2009) note that local knowledge about the absence/lack of desired forest products and services assisted in describing the level of forest degradation in India.

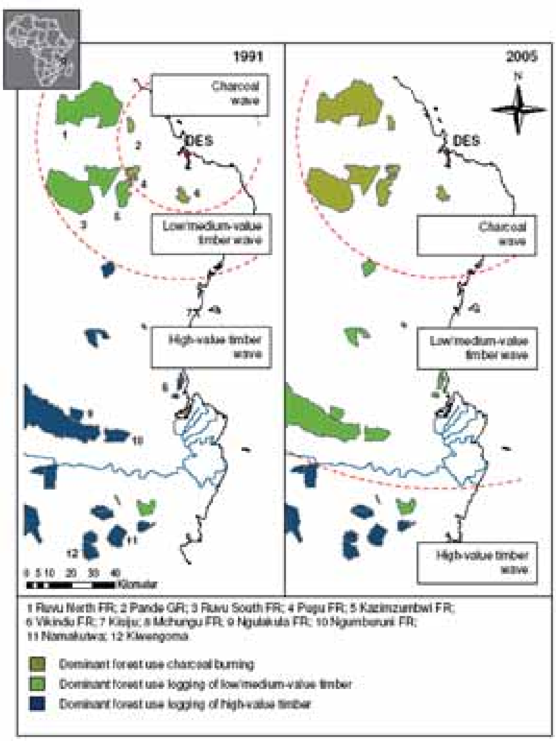
Over-harvesting of timber

In those FMUs subject to a management plan prepared on the basis of sustainability criteria, detecting the over-harvesting of timber consists of comparing the actual level of harvesting (including losses resulting from collateral damage) against the level specified in the management plan. This is usually referred to as the annual allowable cut (AAC); the method for calculating this value in a given FMU is outlined in Box 5.1 (see also 'tree species and ecosystem diversity indicators' in Chapter 4).

In natural (tropical) forest, the essential information needed to determine over-harvesting is the AAC (measured as total volume harvested in a designated area) or allowable harvest intensity (e.g. m^3 per hectare); the minimum dbh, below which a tree may not be harvested (which may vary depending on species); and the net periodic MAI, or net annual commercial increment (usually expressed as m^3 per hectare per year), which is the average annual net change in volume over the measurement period, including growth and recruitment and subtracting losses due to mortality. The main limitation in using any method of yield regulation is the availability of reliable historical data on the growing stock. Indicators that can be used as proxies are discussed below.

FIGURE 5.2

A map of 'degradation waves' of forest goods depicting changes from 1991 to 2005 from Dar es Salaam (DES), the capital city of Tanzania



Note: The numbers correspond to sampled forest areas.
Source: Ahrends *et al.* (2010).

BOX 5.1

Yield regulation in natural tropical forests

Yield regulation is the means by which a sustainable yield is achieved. A sustainable yield implies that products removed from a forest are replaced by growth, with or without management interventions such as enrichment replanting and liberation thinning, without a diminution of other forest values.

The AAC is the volume of timber or area of forest that may be harvested per year. It is usually given as an aggregate figure for all commercial species, but in forest management planning it should be specified by species or species group and by stand or harvesting compartment. It should also relate to commercial yield and size limits.

The MAI is calculated using tree-specific growth functions and/or data obtained from inventories. In calculating the AAC in tropical natural forests the MAI for commercial species should be corrected for expected logging damage. Thus, the AAC should be 50–70% of the estimated commercial MAI, depending on observed levels of logging damage; the share increases as the level of logging damage is reduced.

In estimating the AAC, an allowance for within-tree wastage and degrade should also be made. The latter is necessary if the AAC is based on extracted volumes and is likely to result in an additional correction of 50–70%. Thus, Dawkins' (1964) pan-tropical mean estimate of commercial MAI of 1 m^3 per hectare per year amounts to, in practice, a pan-tropical AAC of 0.25–0.5 m^3 per hectare per year measured as logs at the landing or roadside. There is, however, wide variation in MAI by geographic zone and forest type. According to Whitmore (1998), natural lowland rain forest commonly adds 2–3 tonnes per year of dry weight of bole timber, which may increase to 3.6–12 tonnes per year in forest under good silvicultural management. Growth rates also vary depending on the phase of the forest growth cycle and care should therefore be taken in obtaining or interpreting data on biomass increment in natural tropical forests. A failure to allow for varying growth rates according to growth phase is a major weakness of productivity studies in natural tropical forests.

Timber removals

This indicator corresponds to the volume of timber removed from a given forest area. In forests managed for timber, this volume can be compared with the authorized volume to be extracted according to the forest management plan to assess whether over-harvesting has occurred. The comparison is based on extracted commercial volume and can be done at the level of annual harvesting by compartment, for the entire FMU and by species. Two subsets of timber removals should be estimated: the authorized volume, and the unauthorized volume (i.e. illegally harvested timber); the latter can be estimated through successive forest inventories or using remote sensing. Table 5.2 provides additional information on scale, data requirements and methods for this indicator.

Growing stock. Growing stock is the standing volume of all trees above a selected minimum dbh in a designated forest area. It can be measured in terms of stocking density (m^3 per hectare), basal area (m^2 per hectare) or total volume (m^3 in the designated forest area) on the basis of a forest inventory. Estimates of growing stock always include a margin of error and can be prepared for commercial/potentially commercial timber species or for all species (the latter is needed for the quantification of carbon pools). Chapter 3 explores the measurement of growing stock and its use in assessing forest degradation.

Total growing stock is strongly correlated to forest area; therefore, if forest area declines, so usually does total growing stock. Growing stock per hectare provides a better indication of whether forests are becoming more or less well stocked and therefore may be a better indicator of forest degradation: a decline in growing stock per hectare may indicate degradation (possibly due to over-harvesting) and an increase may indicate recovery from a degraded state and/or a

TABLE 5.2

Measurement of forest degradation based on the supply of forest goods

Goods and services	Scale	Unit of measurement	Method	Comments
Wood removals	Measurement at the stand/FMU/landscape/subnational/national levels. Reported by forest class or type at the administrative unit level (e.g. province, district or municipality) and at the national level	m^3	At the stand level, comparison of volume specified in approved management plans with reported timber removals. At higher levels (e.g. national), comparison of removals with (national) AAC	Illegally harvested wood is excluded and should be estimated by remote sensing or other data
Growing stock for selected species		m^3	Forest inventory, pre-harvest inventory of selected species/tree census	Degradation assessment requires periodic data over time
Fuelwood		m3 or tonnes	Forest inventory, production and consumption surveys	Illegally harvested wood is excluded and should be estimated by remote sensing or other data
NWFP production	Measurement at stand level (e.g. for bamboo and rattan), as well as at FMU and subnational/national levels	m^3, tonnes, number of products collected, etc.	Resource assessments, production records, production and consumption surveys	Degradation assessment requires periodic data over time

reduction in harvesting. It may also be possible to establish minimum 'non-degraded' values for stock density by forest type and geographic zone to serve as a reference point against which the current volume of growing stock may be assessed. This should be based on scientific data supported by expert views, as the results can have strong implications for future management. Such an approach is based on, and requires data from, large-scale forest inventory (see, for example, Bahamóndez *et al.*, 2009).

Damage to the residual stand. The extent of the damage to the residual stand caused by timber harvesting depends on the harvesting techniques used, the skill with which they are applied and the number of trees (rather than the volume) extracted. At a certain point the damage is sufficient to cause deterioration in forest production. Two possible indicators of this are the extent of damage to the residual trees in the forest stand, particularly those of commercial species; and the proportion of ground area disturbed by heavy machines during harvesting. A third measure might be the density of advanced growth and newly established seedlings of tree species, including those of commercial interest.

The collection of data that could be used to measure these indicators may be required by forest authorities but in any case it should be specified in the forest management plan and the monitoring of damage should be part of the annual operational plan. From the perspective of the future supply of timber from a forest, the assessment should provide information on how damage may affect the re-growth of commercial species.

Over-harvesting of individual timber species. Over-harvesting can lead to the depletion of a particular commercial species or species group, especially in tropical forests. In some places, the quest by loggers for certain high-value timber species can be a catalyst for widespread forest degradation (Box 5.2). A measure of the over-harvesting of individual timber species is the ratio of the harvested volume of the species of interest to the authorized (or sustainable) volume.

BOX 5.2

The logging of ipê as a catalyst of forest degradation in the Brazilian Amazon

Ipê (species of the genus *Tabebuia*) is a high-value Amazonian timber. In recent years logging operations targeted at ipê have spread from the eastern Brazilian Amazon, where stocks are exhausted, to the central and western Amazon. Left to market forces, logging is likely to continue to expand. As logging roads and sawmills penetrate new regions they provide access and incentives for colonists and land speculators to follow. This dynamic has been well documented in the Amazon for mahogany, which served as a catalyst for the process of land-clearing, serial logging and burning in the 'arc of deforestation', creating a landscape in which islands of degraded forest (i.e. heavily logged and in many cases repeatedly burned) persist precariously within a matrix of used and abandoned pastures and agricultural fields. The implications of uncontrolled logging of ipê extend beyond the potential depletion of commercial stocks of *Tabebuia* species and could undermine government efforts to achieve forest conservation by bringing order to the Amazon frontier.

Source: Schulze et al. (2008).

Given the lack of up-to-date landscape-scale and national-scale forest inventories at the species level, expert assessments have proved to be cost-effective for well-known, commercially valuable species, such as mahogany (*Swietenia macrophylla*) in tropical America (Kometter, 2004; Grogan *et al.*, 2010). Where possible, however, expert assessments should be supported by periodic inventory data, which are more accurate (albeit much more costly) for establishing sustainable harvesting volumes. In the case of mahogany, the methodology consists of four broad steps:

1. defining 'sampling units' at the national level, combining the known range of the species (through herbarium and permanent-plot data) with current forest cover obtained from satellite images;
2. developing an expert questionnaire to be used in assessment at the level of the sampling unit (containing several FMUs);
3. stratifying density values of commercial-sized trees. A sampling unit with an average (expert-assessed) density of zero commercially-sized trees was considered commercially depleted;
4. presenting the results in national maps showing the sampling units and highlighting contrasting zones of 'commercial viability' and depletion levels.

This approach requires reliable images or maps of forest cover; the availability of experienced field experts; and knowledge of the species' ecology and dynamics for (among other things) determining appropriate minimum harvesting diameters.

In addition to expert surveys, national forest inventories can be used for assessing over-harvesting at the species level. An example is the assessment of density estimates of commercial populations of high-value *Tabebuia* species in the Brazilian Amazon, where forest-cover and geographical distribution data were also used as inputs.

At the operational level, the harvesting intensity of individual species can be managed to avoid over-harvesting based on pre-harvest tree censuses. The enumeration of each tree in the harvesting area and the mapping of its exact location on tree maps provide additional information for operational planning, silvicultural decisions and degradation assessments.

Over-harvesting of NWFPs

The huge diversity of species in tropical forests and the uses to which they are put makes the development of degradation indicators based on NWFPs a difficult task. Nevertheless, it may be possible to identify key areas where NWFP collection is concentrated or NWFPs that are under particular pressure, the monitoring of which may provide suitable indicators of degradation.

Countries provide estimates of the consumption of commercially important NWFPs for various international processes (e.g. for FAO's Forest Resources Assessment). However, such estimates are frequently considerable under-estimates, partly because in many cases the NWFPs are consumed locally and therefore not recorded in official production or trade statistics. For some NWFPs (e.g. resins or barks) it is difficult to determine whether they were harvested in forests or from trees outside forests. Nevertheless, quantifying NWFPs at the FMU level is sometimes possible.

Harvesting intensity vs rate of production. Over-harvesting at the stand level occurs when the annual rate of harvest is higher than the annual rate of production. The annual rate of production can be expressed as follows:

$$P = (X \times Y)/r$$

Where
P = the annual rate of production
X = the abundance of the NWFP stratified by size class (minimum harvestable size)
Y = the average yield per individual of minimum harvestable size
r = the time required to recover to harvestable levels.

If the intensity of harvesting in a given year over a given area exceeds the value of P then the resource is being over-exploited. Ideally, the harvesting rate should be set at a level below that of P. Comparing the annual rate of production with the annual rate of harvest as a robust indicator of over-exploitation is applicable particularly when locally derived data are reliable. Participatory approaches in which NWFP producers and collectors record harvested volumes and recovery times have produced accurate results (see Evans and Guariguata, 2008; Lawrence *et al.*, 2008). A critical assumption is that the annual rate of production does not change from year to year.

In a hypothetical example adapted from Stockdale (2005), assume that all commercially harvestable stems of rattan clumps (climbing palms of the genus *Calamus*) in a secondary forest are cut in one harvest (leaving only immature stems and young resprouts). Yield assessments determine that harvestable stems average 30 linear metres per clump and that a total of 1500 clumps are distributed over the management area. If the time required for the clumps to reach harvestable size has been determined to be four years, then P = (30 x 1500)/4 = 11 250 linear metres per year. A sustainable harvest intensity can thus be set at, for example, 11 000 linear metres per year.

Changes in plant population structure. Another way of assessing the over-exploitation of NWFPs is to determine the shape of size-class distributions of NWFP plant species at the stand or landscape level (see, for example, Peres *et al.*, 2003 for Brazil nut in Brazil, Peru and the Plurinational State of Bolivia). Such information may be obtained from NWFP collectors and producers or from NWFP resource assessments (either specific assessments or as part of forest inventory data collection). A decline in the number of individuals of a particular NWFP species in juvenile size classes may be taken as an indication of a degradation trend in the resource base. It is critical, however, to define the cut-off for the minimum size class. Peres *et al.* (2003) concluded that some Brazil nut populations in parts of the western Amazon lacked juvenile, pre-reproductive trees (≥ 10 cm dbh), suggesting that they were on the verge of collapse under continued harvesting. By including a smaller tree size cut-off (≥ 1.5 m tall but < 10 cm dbh), however, other authors reached the opposite conclusion (Wadt *et al.*, 2008).

A proxy indicator of changing size-class distributions would be the change in the distance travelled by NWFP collectors from their bases over time. Such information could be obtained from collectors.

Changes in production and use. In the context of informal or subsistence harvesting, a possible indicator for measuring forest degradation due to over-harvesting is to use NWFP yield or consumption volume reported in national statistics and compare these over time. Due to a lack of adequate information, however, this approach can rarely be applied in tropical countries (Shackleton, Shanley and Ndoye, 2007).

Changes in consumption. A reduction in the consumption of NWFPs may indicate a reduction in supply. Data on household consumption can be collected using rapid rural assessments such as those specified in the Poverty–Forest Linkages Toolkit.[8] The methodology requires:

- the identification of forest-dependent households and sampling strategy;
- the use of participatory approaches and a random sample of households to ascertain the value of relevant parameters;
- if necessary, the repetition of the survey during different seasons to capture seasonal variations over the course of the year;
- repetition of the procedure at selected intervals to establish change.

In many tropical countries, especially in Africa, significant quantities of NWFPs are sold in local rural markets. The monitoring of these markets can provide additional data on consumption.

Where official data on NWFPs are confined to export markets, baseline surveys among traders and processing plants of NWFPs can provide estimates of internationally traded volumes and changes in demand over time, which can also be used to detect changes in overall production and consumption.

Other social indicators at the local level that could provide indirect information on changes in the supply of NWFPs include average revenue from the sale of specific NWFPs; the average time taken or average distance travelled to collect a specified volume of certain NWFPs; and the importance of specified NWFPs in household resource portfolios. Obtaining information on these aspects usually requires either special periodic surveys or participatory recording by households.

Over-harvesting of fuelwood

National-level forest statistics on the production and consumption of fuelwood and charcoal tend to be gross estimates based on demographic data and estimated consumption per household. Direct measurement of the resource and supply are likely to be more accurate, however, and should be encouraged, particularly in regions with dry tropical forests, where fuelwood and charcoal may be the principal use of wood. At the level of provinces, municipalities or other

[8] www.profor.info/profor/node/103.

subnational administrative units, the supply of fuelwood can be determined by quantifying above-ground forest biomass and its annual increment. Above-ground biomass can be assessed using data obtained from permanent sample plots using allometric models such as:

AGB = SV x WD x BEF

Where
AGB = above-ground biomass
SV = stem volume
WD = wood density
BEF = biomass expansion factor (the ratio of total aboveground biomass to merchantable/usable biomass).

To use this formula as an indicator of degradation, a statistically valid relationship needs to be developed between the above-ground biomass and its increment (i.e. the rate of increase in biomass volume). For each forest type, above-ground biomass increment per hectare multiplied by forest area for each forest type equals annual biomass increment for that area. This value can be compared with annual consumption, stratified by forest type as appropriate.[9]

[9] The original above-ground biomass vs above-ground biomass increment relationship was devised by Clark et al. (2001) and applied at the subnational level by Top, Mizoue and Kai (2004). A re-assessment of the method can be found in Top et al. (2006).

6. Soil erosion

The presence of soil erosion in forests is a prime indicator of forest degradation. Soil erosion can have a major impact on a range of forest services – it reduces water quality, pollutes watersheds with nutrients and sediments, and is an indicator and cause of reduced soil fertility (and potentially, therefore, reduced forest productivity). In an extreme form it can also restrict access to the forest and hinder the extraction of products such as timber. The importance of limiting soil erosion under SFM was reinforced by the inclusion of erosion in Criterion 4 ('conservation and maintenance of soil and water resources') of the Montreal Process Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests. Nevertheless, there have been relatively few attempts to observe and measure soil erosion systematically in forests and in particular to develop a field-usable yet scientifically robust set of methods for describing the various types of erosion, scoring the degree of negative impact of each type, and estimating the quantities of soil lost and impacts on productivity, other ecosystem services and resilience. The aim of this chapter is to provide such a set of methods.[10]

Forest lands tend to be inherently more susceptible to land degradation, including soil erosion, than most arable agricultural lands and rangeland/pasturelands. In most countries, early settlers and developers took the most fertile and flatter lands for agriculture and left the shallow-soiled, rocky, steep, windblown and otherwise low-fertility lands for forestry. The soils of many forests, therefore, are inherently fragile and need ongoing protection against erosion and other forms of degradation. This need may be increased by projected changes in weather patterns due to increasing atmospheric concentrations of greenhouse gases. In most regions there has been an apparent recent increase in the severity of rainstorms, possibly due to long-term climate change. High-intensity, high-volume rainstorms greatly exacerbate soil erosion, particularly of cleared or under-vegetated sloping lands.

Two 'growth areas' of forest use – biofuel production and carbon sequestration (also known as carbon capture and storage) – are further increasing the susceptibility of forests to erosion. Biofuel crops involving trees and perennial shrubs tend to be planted on forest land or 'unoccupied' marginal land, both of which are inherently vulnerable to soil erosion. To ensure rich carbon stocks for maximum sequestration, land fertility and stability are paramount; hence, soil erosion must be stopped or greatly diminished to maximise tree productivity.

Although discussed here only briefly, forest managers should aim to develop an understanding of the links between various types of erosion (and their state, extent and severity) and management activities. Understanding such links will help in developing strategies to repair existing erosion and to avoid or minimize erosion in the future.

There is considerable interaction between erosion and other types of land degradation that, alone and in combination, can have a negative impact on the productivity of forests and previously forested land. For example:

[10] This chapter has been compiled using two main sources of information: an early version of a local-level manual for assessing dryland degradation (FAO, 2011), which derived erosion concepts and indicators from Stocking and Murnaghan (2001), and a report of a project on sustainable pasturelands in Tajikistan by Mulder and McGarry (2010).

- Eroded soil has a reduced nutrient and organic matter content because the soil lost from a site contained nutrients and organic matter.
- The materials remaining on an eroded site are commonly the deeper soil layers (e.g. the B and C horizons), which have inherently lower fertility than the original topsoil.
- Eroded soils tend to have a higher density as a result of lower organic matter status, exposure to raindrop splash, which causes crusting, and the exposure of deeper subsoil layers that are inherently denser. Increased soil density causes problems for seedling placement, germination, vigor and root penetration.
- The greater density and reduced structural stability of eroded soils (due to reduced organic matter) cause eroded sites to be more vulnerable to further erosion.
- In general, the methods presented here are designed to be used in the field by forestry professionals, commonly with the assistance of local people (e.g. land users, forest owners and local forest officials) to aid the interpretation of observed erosion features. While it is written as a stand-alone reference, the methods presented herein require training before they can be used efficiently and effectively.

The prime aim of this chapter is to set out globally applicable, field-usable methods for obtaining qualitative and quantitative information about soil erosion in forests. It can be used to:

- produce a systematic survey of the major erosion features affecting an area as the basis for recording erosion status in the area and comparing between sites that differ in soil, climate and management practices;
- identify the main causes of erosion in an area with a view to
  - understanding the state of erosion through observations of local causative factors
  - recognizing that it is the interaction and summation of such causes that lead to soil erosion
  - identifying potential interventions to repair the erosion features and/or initiating improved land practices to diminish or prevent the erosion process;
- initiate monitoring of the status of erosion features by repeating observations and measures over time. This may be under a non-intervention scenario in which continued degradation is expected, or under an intervention scenario in which management actions lead to improved status;
- for new areas being opened up to forest production or for other land-use changes, record and understand current erosion status and processes to enable management planning that minimizes erosion issues.

WHAT TO MEASURE

Soil erosion occurs when wind and water translocate soil particles. It is exacerbated by poor land management practices, such as the inappropriate placement of roads or timber extraction methods, especially in areas prone to soil movement, such as steep slopes or where there is loose or bare soil.

- Water erosion can be defined as the water-induced detachment and downslope transport of soil particles.
- Wind erosion is the detachment and transport of soil particles by wind action. The assessment of wind erosion commonly involves descriptions and measures of its *impacts*, such as the shapes and dynamics of deposits of particles once they have slowed or stopped, and the effect of the abrasive action of soil particles as they are transported.

This chapter is more concerned with erosion caused by water than that caused by wind. It provides a set of relatively simple, field-usable indicators and measurements to observe, quantify and report on soil erosion through the action of water in forests or on recently deforested land.

Specifically, the methods aim to achieve clarity and uniformity in recording visible soil erosion features, in terms of three distinct but interrelated qualifiers and quantifiers:

- field observations that indicate specified and described types of erosion and the recording of these in terms of four descriptors of the erosion feature – type, state, extent and severity;
- a field scoring method, based on the four descriptors, that provides a quantified basis for inter-site comparisons; and
- field measurements of specified dimensions of erosion types to quantify the rates and quantity of soil loss in a study area.

Selecting observation sites

The selection of observation sites involves at least a two-part process:

- seek sites that are representative of the various land uses in the area under consideration (e.g. cropping land, forest, pasture or horticulture);
- work with local people (e.g. villagers, farmers, herders, forestry workers and farm managers) to identify areas where an understanding of the causes, extent and severity of erosion is needed in order to improve forest and land management or to institute rehabilitation measures.

A ‘desktop’ study should be conducted within the intended study areas to elucidate the major erosion features, their place in the landscape, their elevation and steepness (slope), and their association with the recognizable land uses in the area. This exercise could involve the use of topographic and cadastral maps, Google Earth®, air photos, satellite imagery, digital elevation models, previous reports and soil/geology maps. It should be supported by on-site ground-truthing, including discussions with local people.

Information should be collected at time scales relevant to soil formation and erosion processes. For example, sheet wash may be an annual or more frequent occurrence and rills may form after a series of heavy rainfall events on ploughed land. Gullies and ravines are most commonly the outcome of several seasons or years of concentrated water flows. Repair strategies, therefore, must be prepared and designed for parallel time scales. Rills may easily be ploughed out and can be prevented by the maintenance of appropriate vegetation cover, but it could take years to stabilize and rehabilitate gullies. Local knowledge can be used to cross-reference the observed types, extent and severity of erosion features with recent and historic land practices and weather observations – particularly rainfall periodicity/intensity for water erosion and wind direction/intensity for wind erosion.

Starting observations

Produce a sketch, annotated with observed features, of the area to be evaluated (Box 6.1).[11] This exercise is best conducted by or with the assistance of local stakeholders (e.g. farmers and other land users, foresters and government workers) and is sometimes called a ‘community map’. A typical procedure is as follows:

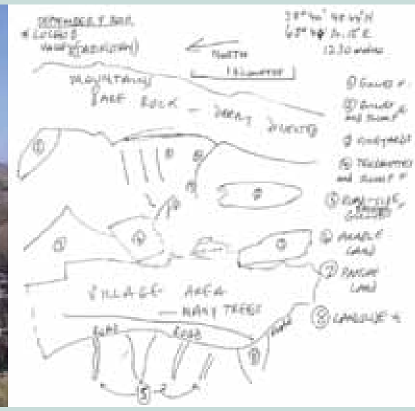
- View the area (e.g. from a distant observation point or, in the case of denser forests, by walking transects and recording information *en route*) and decide the major visible landscape elements to be evaluated (e.g. in terms of slope, land-use type, soils/geology, extent of degradation and conservation features).

[11] Note that this sketch, along with all information on the four erosion indicators, should be recorded on the field worksheet (Annex 5), to provide uniformity of data collection and reporting as well as a lasting record of information collected at all sites, to aid the monitoring of erosion change over time or with change in practice.

BOX 6.1

Community mapping

The photo shows a distant view of an area of land north of Dushanbe, Tajikistan, that is to be investigated for erosion features. The community map was produced on the basis of this view, with discussions with local stakeholders helping to delineate and describe the main erosion features and to provide a range of other relevant information (such as vegetation types, main land uses, slopes, villages, roads and streams). The location of the observation point (latitude, longitude, elevation and north point) was obtained using a global positioning system.



- Sketch the various landscape elements.
- Indicate the proportion of the total land area they occupy.
- Number (or annotate) the various landscape elements and make a separate description for each sub-area in terms of the following
  - land use – e.g. villages, pastures, vineyards, orchards, forest and streams/rivers.
  - erosion features – e.g. sheet erosion, rills and gullies – and any activities with which they seem to be associated, such as newly planted forest, recent timber extraction, clearings and roads. Annex 3 and Annex 4 describe and illustrate (respectively) the major types of erosion.
  - any existing conservation/sustainable land management measures.

In discussion with local stakeholders, obtain the history of land use (e.g. how long it has been forested or deforested, previous land uses and management practices, the species being grown and for what purposes); recent and past significant weather (e.g. intense rain events, flash floods and greater-than-average rainfall); and perceptions of erosion and current land productivity and how these might have changed (e.g. "the land used to produce larger and better trees", "now with every rain event we lose more soil", and "the streams are full of soil after every rain"). The outcomes of such discussions can be used to interpret and explain the causes of existing erosion features.

Type, state, extent and severity

Once the main erosion features (as observed from a distance) have been drawn on the community map, each area of soil erosion can be qualified in terms of four descriptors: type, state, extent and severity, as set out below. Although there are shortcomings in some of the classes and terms used for these descriptors, this should not prevent their use, particularly since their widespread application will help to improve their definition.

Type. Table 6.1 presents the 13 types of erosion feature. The methodology for describing erosion types progresses from those that are least evident in the landscape (e.g. rain-splash and sheet wash) to those that are most evident (e.g. ravines and landslides). Note that 'type', as used here, describes only the physical nature of an erosion feature. To best distinguish between the end-point of one type and the beginning of another (e.g. when does a rill become a gully?) in the field, each feature requires both descriptors of its physical nature and information on its physical boundaries.

TABLE 6.1

Definition and scoring for erosion types

Soil erosion type (score)	Code	Definition	How to recognize
1 Splash (1)	**SP**	Raindrop impact displaces soil particles vertically and downslope	Soil particles on lower parts of plants and/or a compacted (or dispersed) soil surface crust
2 Sheet wash/sheet (2)	**S**	Erosion of the top layer of the soil (also termed mudflow), as opposed to linear erosion (such as rill, gully and ravine)	Gravel/stones protruding from soil surface; root exposure; loss of darker topsoil horizon; subsoil exposure
3 Rill (2)	**R**	Irregular, downslope linear channels, up to 0.3 m deep and wide	Shallow, commonly long channels running downslope
4 Gully (4)	**G**	Irregular, v-shaped, steep-sided linear channel formed in loose material, 0.3–2.0 m deep, formed by water erosion	Deep pronounced channels
5 Ravine (4)	**A**	As per the definition of 'gully' but very deep and wide (i.e. > 2 m deep and wide)	Very deep and wide channels
6 Landslide (4)	**L**	Sudden downslope movement of a concentrated mass of soil and rock, triggered by water saturation or earthquake (sometimes termed mass movement)	Almost vertical sides; rounded head (gully has narrow or sharp head)
7 Slumping (2)	**SL**	Slow, irregular, downward progression of a thin (< 1 m) layer of soil, due to water saturation, possibly in combination with freezing–thawing process	Rounded scar; irregular, uneven downslope surface
8 Rotational slumping (3)	**RS**	A form of mass movement where rock and soil move downward along a concave face (see also 'slumping'). The rock or soil rotates backward as it moves in a rotational slip. Rotational slumping differs from a landslide as it always has a concave sliding plane and multiple scars; landslides have relatively straight shear planes	Series of irregular scars and wide cracks
9 Terracette (2)	**T**	Small, irregular step-like formations due to a combination of slumping and preferential animal movement (tracks) on the surface of moderate-to-steep slopes	Irregular on contour steps of about 0.1–0.2 m height on moderate-to-steep slopes in grasslands
10 Tunnel (3)	**TU**	Often hidden subsurface holes and tunnels that can break through to form surface gullies	Often hidden but may break through the soil surface as potholes and gullies
11 Roadside erosion (2 or 3)	**RE**	Erosion (mostly gullies) caused by concentrated water flow over an impervious road surface, cutting back into the road and damaging the road or eroding downslope. Score depends on gully or tunnel intensity	Erosion features below the point where water runs off the road
12 Streambank erosion (2 or 3)	**SE**	Undercutting of streambank by running water. Score depends on gully or tunnel intensity	Fresh cuts in banks; exposed tree roots; collapsed structures
13 Wind erosion (variable)	**WE**	Detachment and transport of soil particles by wind. Scoring is difficult because observed features (e.g. dunes and the scouring of vegetation and fence posts) are almost always the effects of wind erosion	Scouring on windward side; deposits at leeward side of obstacles; sand dunes
Note: Annex 3 contains additional descriptions of most of these erosion types, and Annex 4 shows examples of them.			

State. One of four classes is used to indicate the activeness of an erosion feature:

- active – it is increasing in size or extent;
- partly stabilized – it is between active and stable;
- stable – it is either a historic (relic) feature caused by climatic events or land uses, or a more recent erosion feature, the activity of which has ceased as a result of management interventions (e.g. the installation of contour banks);
- decreasing – recent management interventions have begun to reverse the erosion process (e.g. the filling of gullies by rock and vegetation has helped stabilize and hold soil).

Extent. The intent here is less to measure the actual area of an erosion feature (although this can be done) and more to estimate the proportion (expressed as a percentage) of the landscape affected by a particular erosion type.

The five terms used to define extent are:

- negligible (0–2% of the area under study);
- localized (3–15% of the area);
- moderate (16–30% of the area);
- widespread (typically 31–50% of the area).

Note that the class 'widespread' is intentionally only ranged up to 50% of the area. This reflects that each erosion type is classed individually, so it is possible (in one area) that there is each of (for example) sheet wash, terracettes and gullies, with the extent of each being localized (10%), widespread (50%) and moderate (20%), respectively – showing that 80% of the area is eroded but with a range of types.

There are various ways to record extent. The areas affected by a given erosion type can be drawn on a community map. Where available, erosion features can be identified or drawn onto, for example, maps, air photos, orthophotos, satellite images or Google Earth® images.

If required for detailed study, a theodolite or dumpy level can be used for the accurate mapping of recorded erosion features, although this requires a relatively high level of skill and access to relatively expensive equipment.

Severity. Erosion severity is generally defined as the 'degree of the effect of the (specified) erosion type'. A more pragmatic definition is the rate or 'average amount of soil that is moved by water or wind' (Leys *et al.*, 2010), expressed as mass per area over a specified unit of time. Based on this latter definition but recognizing that the mass of soil loss will rarely be known (particularly for historic erosion features), Leys *et al.* (2010) derived a field-usable estimate of erosion severity using the following five classes:

- low (minimal erosion evident) – most commonly used for splash or rill types;
- moderate (there is evidence of erosion – the surface has been lowered by less than 0.1 m but eroded sediment remains within the area under study);
- high (there is evidence that sediment is being exported off site and the surface has been lowered by less than 0.1 m);
- severe (there is evidence that sediment is being exported off site and the surface has been lowered by 0.1–1 m);
- extreme (there is evidence that sediment is being exported off site and the surface has been lowered by more than 1 m).

By definition, certain erosion types will never be of ’low’ or ‘moderate’ severity. For example, gully, ravine, landslide and tunnel erosion types must be rated either severe or extreme because the erosion feature is >0.1 m deep. Nevertheless, insidious sheet or rill erosion that continues year by year over large areas may be equally serious as, or more serious than, widely spaced gully erosion in terms of total soil loss and impact, especially in shallow soils.

MEASUREMENT METHODS

This section provides two sets of field techniques that quantify soil erosion features. The first is a simple field-scoring system to facilitate comparisons of erosion status and trends between sites that vary in, for example, management, soil and vegetative cover. The second gathers data on quantities and rates of soil erosion.

Under the field-scoring system an erosion feature is allocated a score for each of the four descriptors (type, state, extent and severity, as described above). Table 6.1 shows the scores for a range of erosion types and Table 6.2 shows the scores for state, extent and severity.

Users should be aware of a number of issues associated with this scoring system, including the following:

- The allocation of scores to erosion types is somewhat arbitrary.
- As discussed above, certain erosion types will never be describable as of ‘low’ or ‘moderate’ severity. Thus, the gully, ravine, landslide and tunnel erosion types not only score 4 for type, they also score 3 or 4 for severity.
- If an area is subject to several types of erosion, the current system scores each type separately and sums the individual scores to give a composite score. This is because the various erosion types are likely to be related and to have a summative negative effect on forest condition or productivity. This composite scoring system may change in the future with wider use of the system. Box 6.2 provides some worked examples of this scoring system.

The final score for any given erosion type in a study area, obtained by summing the scores for type, state, extent and severity, indicates the erosion class (Table 6.3).

TABLE 6.2

Scoring for erosion state, extent and severity

State	Score	Extent	Score	Severity	Score
active	3	widespread	3	extreme	4
partly stabilized	2	moderate	2	severe	3
stable	1	localized	1	high	3
decreasing	0	negligible	0	moderate	2
				low	1

TABLE 6.3

Erosion classes derived from summing scores for type, state, extent and severity

	Erosion class				
	negligible or decreasing	low/weak	moderate	severe	very severe
Score	0–1	2–5	7–10	10–12	13+

BOX 6.2

Worked examples of scoring erosion features

Example 1. Gully erosion (score for type = 4) on a site that is actively eroding (score for state = 3) and widespread (score for extent = 3). Gullies are greater than 1 m deep (score for severity = 4). Total (summed) score = 14. Overall, therefore, the erosion class is 'very severe'.

Example 2. Rill erosion (score for type = 2) that is partly stabilized (score for state = 2), localized (score for extent = 1) and of moderate severity (score for severity = 2). Total score = 7. Overall, therefore, the erosion class is 'moderate'.

Example 3. Ravine erosion (score for type = 4) that is decreasing in activity (score for state = 0), moderate in extent (score for extent = 2) but severe (score for severity = 3). Total score = 9. Overall, therefore, the erosion class is 'moderate'.

Example 4. The area has two erosion types: splash (score for type = 1) that is active (score for state = 3) localized (score for extent = 1) and of low severity (score for severity = 1) – total score = 6; and landslide (score for type = 4) that is stable (score for state = 1), localized (score for extent = 1) and extreme (score for severity = 4) – total score = 10. Total score (i.e. sum of the two erosion types) = 16. Overall, therefore, the erosion class for the area is 'very severe'.

Example 5. The area has three erosion types: sheet wash (score for type = 2) that is active (score for state = 3), localized (score for extent = 1) and of moderate severity (score for severity = 2) – total score = 8; terracettes (score for type = 2) that are active (score for state = 3), localized (score for extent = 1) and of moderate severity (score for severity = 2) – total score = 8; and gullies (score for type = 4) that are partly stabilized (score for state = 2), localized (score for extent = 1) and extreme (score for severity = 4) – total score = 11. Total (i.e. sum of the three erosion types) score = 27. Overall, therefore, the erosion class for the area is 'very severe'.

Note that while the scores obtained by this method offer some basis for comparing the impact between erosion features and/or across sites, it is inherently difficult to make definitive comparisons between such physically different types of erosion as rills and gullies. An entire landscape may be subject to rill erosion and the resulting soil loss may be very large, with important implications for soil fertility and productivity. On the other hand, a few large ravines in the same area would pose very different management problems (e.g. access for management interventions and harvesting) and would possibly require major, expensive interventions to repair and conserve. Moreover, while sheet and rill erosion generally have relatively low scores under this methodology, their cumulative effects should not be underestimated, particularly as they can strip away surface soil layers that are generally richer in organic matter due to litter accumulation and are vital for continued site productivity.

The second set of measurement methods involves estimating the volume of the space from which the soil has been removed by erosion (assumed to be equivalent to the volume of removed soil) and relating this, where possible, to known time scales of erosion activity. Direct methods for measuring soil loss in the field are applicable only to three of the 13 erosion types in Table 6.1 – rill, gully and ravine. Indirect methods based on measuring the *effects* of erosion are required for the other erosion types in Table 6.1.

Direct measures[12]

Rills. The measurement of soil loss from rills assumes that the depression forms a regular geometric shape that is estimated to be triangular, semi-circular or rectangular in cross-sections, as determined by field observation.

To estimate the quantity of lost soil, measurements are made of rill depth, width and length. It is important to collect a number of measurements of both the width and depth of any one rill, and to measure many rills in the study area, to obtain an average cross-sectional area. The average rill catchment area – that is, the area of land that contributes soil-laden water to a rill – must also be estimated for the rills in a given area. If it is known how long the rills in an area have taken to form (if, for example, the land was last cultivated two months or two years ago, or has only recently been cleared of forest), an annual rate of soil loss can be estimated for that rill.

Using the average measurements of width and depth and an assessment of the cross-sectional shape of the rill, calculate the average cross-sectional area, using the appropriate formula as follows:

triangle cross-sectional area = 0.5 × width × depth
semi-circle cross-sectional area = 1.57 × width × depth
rectangle cross-sectional area = width × depth.

Multiply the cross-sectional area by the length of the rill to estimate the volume of soil removed by erosion. The volume removed per unit area can also be calculated. Box 6.3 provides a worked example of these calculations.

Gullies and ravines. Gullies and ravines have a similar general shape – a flat floor and sloping sides. The bottom of these features (the floor) is less wide than the top (parallel to the soil surface); such a shape best approximates a trapezium (Figure 6.1). The calculation of soil loss is similar to that for rills, except that a different cross-sectional shape is used.

BOX 6.3

Worked example – estimating soil loss in rills

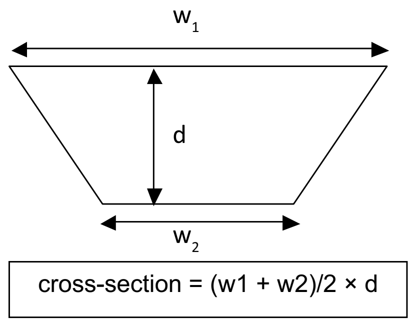
For a case in which the average dimensions of many measured rills are width = 0.12 m, depth = 0.042 m:

1. The average cross-sectional area of the rills in the study area, assuming a triangular cross-section, is: $0.5 \times 0.12 \times 0.042 = 0.00252\ m^2$.
2. Assuming the average rill length in the study area is 2.5 m, the volume of soil lost from an average rill is: $0.00252 \times 2.5\ m = 0.0063\ m^3$.
3. The volume of soil lost from the estimated catchment area (12 m^2) is converted to volume per m^2, as follows: $0.0063/12 = 0.000525\ m^3$ per m^2.
4. The volume per m^2 is converted to tonnes per hectare, using an estimated soil bulk density value of 1.3 tonnes per m^3, as follows:
   $0.000525 \times 1.3 \times 10\,000 = 6.9$ tonnes per hectare.

In this example, therefore, 6.9 tonnes per hectare of soil have been lost due to rill erosion.

[12] This section is based on the original concepts of the quantification of field-observed erosion features in Stocking and Murnaghan (2001).

FIGURE 6.1
Calculation of the cross-section of the trapezoid shape of gullies and ravines



To calculate the quantity of soil lost through gully or ravine erosion, measure the feature's depth, width at lip (i.e. the top) and base, and length. This can be done with a laser-based rangefinder or a 30 to 100 m tape. A number of measurements of both width and depth should be made along the length of any one feature, and many similar features in the study area should be measured, to achieve a representative sample.

The annual rate of soil loss is estimated more easily for gullies and ravines than it is for rills, as the former are more-or-less permanent features of a landscape. The necessary information can be obtained in various ways, including through repeated visits (particularly if permanent monitoring stakes are installed as reference points) and from time series of aerial photographs and/or satellite imagery. Even with such methodologies, however, the annual rate of soil loss is at best an approximation due to factors such as the varying rates of soil loss that can occur as the gully or ravine deepens and different layers of soil are exposed; variation in rainfall totals and periodicity; and changes in tree density (e.g. due to harvesting and silvicultural operations such as thinning) and hence runoff rates over time. Tunnelling may also occur on the sides of gullies and ravines, greatly exacerbating soil loss in some years.

To calculate soil loss from gullies and ravines, use average measurements of the widths at lip and base as well as the feature's depth to calculate the average cross-sectional area of the feature, using the following formula:

(width at lip + width at base)/2 × depth

Box 6.4 provides a worked example of these calculations.

Indirect measures

Indirect measurements of soil erosion rely on features observed and measured in the field that demonstrate the effects of soil erosion, referred to here as erosion proxies. In total, seven erosion proxies are presented: plant root exposure; exposure of the bases of fence posts and similar structures; tree mounds; pedestals; solution notches and rock coloration; armour layer; and soil build-up against a barrier. Erosion types that most commonly lead to these erosion effects are rain-splash, sheet wash and wind erosion.

With all but the last of the seven proxies (i.e. soil build-up against a barrier), the general approach to estimating soil loss is to measure the current (eroded) soil level against the evident location of the original (or at least a recent previous) topsoil level. Particularly in terms of

BOX 6.4

Worked example – estimating soil loss in gullies and ravines

For a case in which the average dimensions of many measured gullies or ravines are – width at lip = 10.2 m; width at base = 4.8 m; depth = 2.0:

1. The average cross-sectional area of the gullies and/or ravines in a study area, assuming a trapezoidal cross-section, is:
   $((10.2 + 4.8)/2) \times 2.0 = 15\ m^2$.
2. Assuming the average gully and/or ravine length in the study area is 200 m, the volume of soil lost from an average gully or ravine in the study area is:
   $15 \times 200\ m = 3\ 000\ m^3$.
3. The volume of soil lost from the estimated catchment area (1 km^2) is converted to volume per m^2, as follows:
   $3\ 000/1\ 000\ 000 = 0.003\ m^3$ per m^2.
4. The volume per m^2 is converted to tonnes per hectare using an estimated soil bulk density value of 1.3 tonnes per m^3, as follows:
   $0.003 \times 1.3 \times 10\ 000 = 39$ tonnes per hectare.

In this example, therefore, 39 tonnes of soil per hectare have been lost due to gully or ravine erosion.

measuring soil loss against living objects such as trees or other plants, if the planting date is known then an estimate of annual soil loss is possible. The same is true if the date of the installation of fences, poles, walls, houses and other potential proxies is known.

In measuring soil build-up against a barrier the reverse is measured – that is, the accumulation of eroded sediments behind a physical barrier such as a hedge or fence. The depth of this deposited soil is measured relative to the current topsoil level. The amount of soil loss can only be estimated if the area contributing eroded material and the area of deposition can be determined.

Plant root exposure. The removal of soil particles by water or wind can lead to the exposure of plant roots as the overall soil level decreases. Close inspection of the lower portion of a tree trunk or other plant stem may reveal a mark indicating the level of the original soil surface. By measuring (with a ruler) the vertical difference between this mark and the present soil surface, an estimate can be made as to how much soil has been lost.[13] In the case of lateral roots away from the tree trunk, the upper surface of the most exposed roots is usually taken as the former soil surface. For planted forests and perennial and annual crops the soil loss estimate would cover the period since the tree or crop was planted. In natural vegetation it may be less easy to determine the period over which the measured soil loss took place.

A number of exposed plant roots should be measured and the average taken to improve the representativeness of the sample. Results should also be compared with those of other erosion indicators (such as those described below) as a way of cross-checking that they are realistic. Box 6.5 provides a worked example of this method.

[13] The loss of an average of 1 mm of soil across a hectare of land is equivalent to 13 tonnes if the bulk density of the soil is 1.3 g per cm3.

Note the following to help ensure the validity of the data collected:

- Differences in root exposure may reflect different erosion processes (eg rain-splash and sheet wash) taking place in the same area.
- Roots and stems might capture runoff to form erosive channels, thus increasing soil loss. On the other hand they might slow surface flows, allowing deposition to occur, and they might also trap and allow the accumulation of windblown material.
- Some plants have a tendency to lift themselves out of the ground as they grow, which could affect estimates of soil loss based on measures of plant root exposure. This effect may be observable, however, especially in stony soils where larger platy fragments occur. Assessors should look for evidence of plant lift in the alignment of stones, as growth may force a rearrangement of stones so that they become tilted, with the raised end nearest the trunk.
- Tree roots may expand in diameter as the tree grows, so that roots running parallel to the soil surface rise to or above soil level, which could affect estimates of soil loss based on measures of plant root exposure.

Exposure of the bases of fence posts and similar structures. Similar to plant root exposure, the exposure of the bases of anthropogenic structures such as fence posts, house and bridge foundations and telegraph poles can indicate soil loss, principally from rain-splash, sheet and wind erosion.

Generally the distance between the new ground surface and the point on the post that would originally have been at ground level is measured using a ruler. Soil loss is estimated in the same way as for plant root exposure (see worked example in Box 6.5).

Note that the age of the structure is required if an annual rate of soil loss is to be estimated. As for plant roots, the structure itself could influence the rate of erosion or deposition, and this should be taken into account. Moreover, the effect of the structure may vary depending on such factors as rainfall amount, intensity and periodicity and wind direction and strength. Local knowledge can help in assessing the effect of climatic variables on soil erosion and accumulation.

Tree mounds. The use of tree mounds to provide estimates of soil loss depends on the raindrop-energy-absorbing properties of tree canopies (their 'umbrella' effect). Because of these properties, soil under a tree canopy can be less eroded than soil in an adjacent area without trees because it has been protected from raindrop impact and subsequent rain-splash and sheet erosion.

BOX 6.5

Worked example – estimating soil loss using plant root exposure

For a case in which the average of measurements of the height difference between the top of exposed plant roots or stem and the current soil surface (i.e. the average depth of soil loss) = 5.88 mm:

1. Convert this drop in soil level to tonnes per hectare, using an estimated soil mass of 13 tonnes per mm per hectare (assuming an average soil bulk density of 1.3 g per cm^3)
   $5.88 \times 13 = 76.4$ tonnes per hectare.
2. If the average age of the plants where the soil-level change was measured was four years, then the estimated annual soil loss is
   $76.4/4 = 19.1$ tonnes per hectare per year.

In this example, therefore, about 19 tonnes of soil per hectare per year were lost to soil erosion.

The difference in height between the soil surface under a tree and in an adjacent exposed area provides an indicator of the amount of soil loss that has occurred during the life of the tree (tree age may be obtained from forest records or by talking to locals). It is recommended that such measurements are recorded for a range of trees of different sizes and ages in the study area, as there is large variation between species in the capacity of canopies to protect the underlying soil (some species may be leafless during the peak rainy season, for example). Soil loss is estimated in the same way as for plant root exposure (see example in Box 6.5).

Note the following to help ensure the validity of the data collected:

- Mounds around the base of trees, shrubs and other plants may have been caused by factors other than erosion (e.g. termites, sedimentation or tree-litter build-up).
- Some trees may lift the soil around them as they grow, thus creating natural mounds. This could affect estimates of soil loss.
- Tree canopy size and density change as the tree grows; thus, a tree mound created due to canopy protection may not develop uniformly over time. Measurements should be taken at various distances from the tree trunk and averaged.

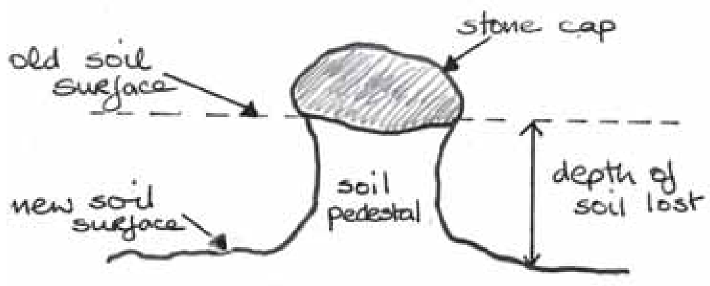
Pedestals. A pedestal is a column of soil that stands out from the general eroded surface, protected by a cap of resistant material such as a stone or root. Bunch grasses can also protect the soil immediately beneath them (comparable to tree canopies and tree mounds, above) to produce a pedestal-like feature. Care is required, however, in interpreting these latter forms.

Pedestals are caused by differential rain-splash erosion, which dislodges soil particles surrounding the pedestal but not under the resistant cap (Figure 6.2). Pedestals can be created artificially by pressing bottle tops into the soil (a technique that can be used for monitoring in areas where rain-splash erosion rates are potentially very high).

Pedestals can be measured using a ruler. A number of measurements should be made in the study area, even to the extent of stratifying the area and averaging pedestal height in each of the strata to account for across-site variability. Assuming that the cap of a pedestal was at the soil surface when erosion started, the gap between the base of the cap and the base of the pedestal (i.e. where it meets the general soil surface) should be measured. This measure represents the soil loss since the soil was last disturbed (e.g. through forest clearing or cultivation). Soil loss is estimated in the same way as for plant root exposure (see example in Box 6.5). If the time of that disturbance is known it is possible to estimate an annual rate of soil loss.

FIGURE 6.2

Sketch of a soil pedestal capped by a stone



Source: Stocking and Murnaghan (2001).

Note the following to help ensure the validity of the data collected:

- Pedestals often form under trees or crops where intercepted rainfall falls to the ground as larger drops. If this is the only location in which pedestals are found they would provide an unreliable estimate of the level of soil loss for a larger area.
- Measurement of pedestals in association with clumps of vegetation should be avoided as vegetation clumps can accumulate soil.
- Capping stones may originally have been buried in the soil and are now exposed with an underlying pedestal. In this case, the pedestal method would underestimate erosion.
- The localized redistribution of material eroded from under the stone needs to be accounted for when estimating soil loss.

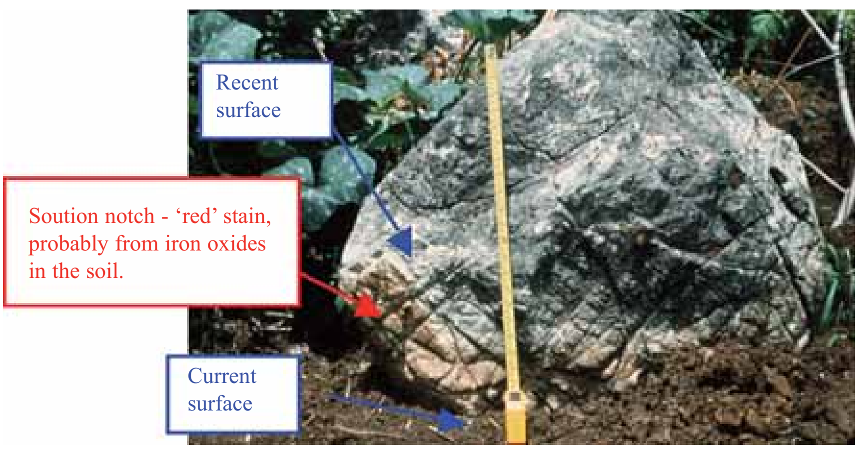
Solution notches and rock coloration. Solution notches are indentations found on rocks that indicate historic soil levels (Figure 6.3). They are created by chemical interaction between soil, air and rock and can mark a previous level of the topsoil where, due to its high organic matter content (and therefore high levels of humic acids), it etched a notch in the rock at the air/soil interface. The discoloration of stones or rocks can also indicate historic soil levels due to similar chemical processes. Solution notches are most likely to occur in limestone and calcareous rocks, which are particularly susceptible to acidic organic chemicals.

To estimate soil erosion, the distance between the notch or coloration to the current soil level is measured using a ruler. A number of measurements should be made in the study area and the average taken. Soil loss is estimated in the same way as for plant root exposure (see example in Box 6.5). The rate of loss is more difficult to assess given the difficulty of determining the time since notching or coloration. Sometimes it may be possible to calibrate the rate using comparisons with other soil-loss indicators (such as plant root exposure).

Armour layer. An armour layer is the concentration, on the soil surface, of coarse soil particles that ordinarily would be distributed randomly throughout the topsoil (Figure 6.4).

FIGURE 6.3

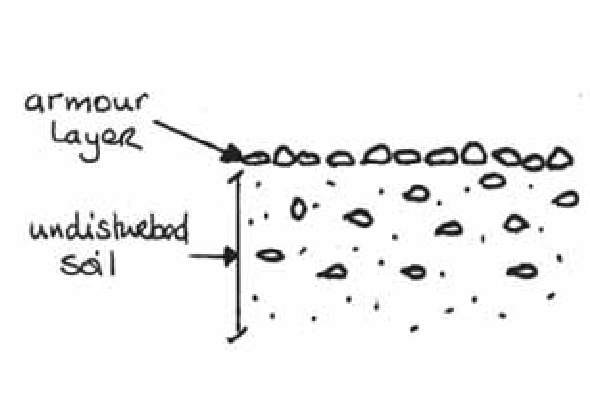
Solution notch and staining in limestone rock



Source: Stocking and Murnaghan (2001).

FIGURE 6.4

Diagrammatic representation of an armour layer and photograph showing removal of a portion of an armour layer



Source: Stocking and Murnaghan (2001).

The concentration of coarse material in the armour layer suggests that finer soil particles have been removed selectively by the energy of wind and/or water, leaving behind heavier particles that are less easily moved. The armour layer can be measured by digging a hole to reveal the depth of the coarse top layer. Several measurements at different places on the site should be made and the average depth calculated. The approximate proportion of stones and coarse particles in the topsoil below the armour layer is judged by taking a handful of topsoil from below the armour layer and separating the coarse particles from the rest of the soil. In the palm of the hand, an estimate is made of the percentage of coarse particles in the original soil. This estimate should be repeated at different points on the site. The depth of the armour layer is compared to the amount of topsoil that would originally have contained a similar quantity of coarse material. The volume of finer soil particles that have been lost through erosion can then be estimated. Box 6.6 provides a worked example of this calculation.

Note the following to help ensure the validity of the data collected:

- Stones may be present on the surface for a range of reasons. For example, they may have been exhumed from the subsurface soil by animals or frost action.
- The precise and accurate measurement of the thickness of the armour layer is critical – every 1 mm of armour is equivalent to a soil loss of 13 tonnes per hectare (assuming an average soil bulk density of 1.3 g per cm^3).
- As well as erosion processes, repeated shallow tilling of the soil may concentrate stones near the surface. Where this happens, estimates of the erosion rate will tend to be exaggerated unless the percentage concentration of stones in the original soil is based on an estimate from well below the (tilled) topsoil.

Soil build-up against a barrier. The build-up of eroded material against a barrier can be used to estimate the movement of soil across the area of interest (rather than the loss of soil from the area). In this case, the eroded materials are halted by an obstruction and the materials are deposited against the obstruction (Figure 6.5). The result is a build-up of soil against the barrier.

BOX 6.6

Worked example – estimating soil loss using armour layer

For a case in which the average of measurements of the thickness of the armour layer = 1 mm:

1. Convert the average soil loss to the equivalent in metres:
   1.0 × 0.001 = 0.001 m.
2. Calculate the depth of soil required to generate an armour layer 0.001 m thick, where the proportion of coarse material in the topsoil was determined to be 20% (i.e. a 1:5 ratio):
   0.001 × 5 = 0.005 m.
3. Calculate the depth (m) of soil lost by subtracting the armour layer from the total depth of soil:
   0.005 – 0.001 = 0.004 m.
4. Convert the drop in soil level to tonnes per hectare, using an estimated soil bulk density of 13 tonnes per hectare.
   0.004 × 13 000 = 52 tonnes per hectare.

In this example, therefore, 52 tonnes of soil per hectare have been lost to soil erosion.

FIGURE 6.5

Build-up of soil behind a *Gliricidia* hedge, Sri Lanka

Note the difference between the level of the soil where the person is standing (A) relative to the build-up on the upper side of the hedge (B).
Source: Stocking and Murnaghan (2001).

The volume of soil trapped behind the barrier can be estimated by measuring the depth of the soil deposited and the area over which it is deposited. Where the build-up is against a continuous barrier such as a fence or hedge the measurement will give an approximation of total soil loss from the soil ‘catchment’. A visual examination of the area close to the barrier will indicate how far the deposition extends into the field. This distance (length) should be measured at a number of points. The depth of the soil accumulated against the barrier can be determined by examining the soil level on the far side of the barrier (point A in Figure 6.5). In order to calculate the total amount of accumulated soil, a linear slope is assumed and the wedge of soil behind the barrier is treated as a triangle. The annual rate of soil loss from an area can be derived by dividing the quantity of accumulated soil by the number of years that the barrier has been in existence. Box 6.7 provides a worked example of these calculations.

BOX 6.7

Worked example – estimating soil loss using soil build-up against a barrier

For a case in which the estimated total catchment area behind the barrier = 70 m^2, the length of the barrier = 7 m, the average depth of the deposit at the barrier = 0.16 m, and the length of the accumulation upslope of the barrier = 0.945 m:

1. Calculate the average cross-sectional area of deposit (assumed to be triangular) using the formula
   0.5 × depth at barrier × horizontal length
   i.e. 0.5 × 0.16 × 0.945 = 0.0756 m^2.
2. The total volume of soil accumulated behind the barrier is calculated using the formula
   Cross-sectional area × barrier length
   i.e. 0.07560 × 7 = 0.5292 m^3.
3. The volume of soil lost per m^2 of catchment can be calculated by dividing the total volume of soil by the total catchment area, as follows
   0.5292/70 = 0.00756 m^3 per m^2.
4. The volume per m^2 can be converted to tonnes per hectare using an estimated soil bulk density value of 1.3 tonnes per m^3, as follows
   0.00756 × 1.3 × 10 000 = 98.3 tonnes per hectare.
5. The barrier was constructed three years prior to measurement. Thus, the annual soil loss represented by the soil accumulated behind the barrier is
   98.3/3 = 32.8 tonnes per hectare per year.

In this example, therefore, there has been a movement of 32.8 tonnes of soil per hectare per year in this area.

Note the following to help ensure the validity of the data collected:

- It is possible that soil on the lower side of the barrier has been lowered by erosion, in which case the measure of accumulated soil on the upper side of the barrier may be overestimated.
- Estimates do not differentiate between sediment produced within the immediate soil catchment and sediment produced further upslope. It is possible therefore that this method could overestimate erosion per unit area.
- Not all materials transported in runoff will be deposited at the barrier. The speed, volume and direction of runoff all influence the extent of deposition. Therefore, estimates of soil loss derived by this method may be underestimates.
- Forest clearing may increase soil depth behind barriers, particularly where conservation techniques such as terracing have been introduced to lessen the effect of slope. If the slope was convex before the barrier was constructed, the estimate of soil loss will be underestimated because it assumes a linear slope.
- The soil level below the barrier may not be the original soil level. In Figure 6.5, for example, the area immediately below the fence has been excavated and levelled for road-building.

MEASUREMENT INTENSITY, FREQUENCY AND REPORTING

It is difficult to be prescriptive about the intensity and frequency of soil erosion monitoring and reporting in forests. The rate and severity of erosion vary greatly depending on climate, soil type, slope, vegetative cover and the nature and intensity of disturbance. Generally, therefore, the frequency and intensity of monitoring and reporting should be set according to circumstances.

An important consideration in embarking on a soil erosion monitoring programme in a forest area is the establishment of benchmarks and monitoring protocols. Thus, the data collected at the commencement of the programme act as the baseline for all subsequent observations and measurements. A repeatable methodology should be used so that, over time, such observations and measurements provide an indication of changes in erosion over time and the possible effects of interventions. In forests, interventions could include the construction of physical or vegetative barriers to mitigate the negative impact of the observed erosion, and thinning, logging and/or clearing, each of which could have significantly different effects on erosion and induce differing erosion rates. Monitoring frequency may need to vary accordingly and should be adequate to capture the effect of the intervention. Commonly, monitoring should be more frequent immediately after an intervention and become less frequent as rates of erosion decline. Where there are few or no significant human interventions, erosion may be monitored at a fixed interval determined on the basis of the intensity of the erosion process (e.g. annually in active erosion situations or in sensitive watersheds, and perhaps at 5–10-year intervals where erosion is considered to be less of a problem).

The intensity of monitoring refers to the number of observations to be conducted at any given time in an area of interest. A prescriptive approach is impossible due to the many situations that may be experienced. Several ‘entry levels’ could be considered, such as:

- At the simplest level, a community map (Box 6.1) could be sketched rapidly at short time intervals and the sequence compared to determine the more active or widespread areas and types of erosion features for closer investigation.
- Another relatively simple entry point would be to describe and class the erosion features present in an area of interest (e.g. on the basis of tables 6.1, 6.2 and 6.3).
- The measurement of soil loss is the most time-consuming and therefore tends to be used less often and less intensively.

The intensity of monitoring is also governed by the types of erosion features that occur in a study area. For example, if there are only 5–10 gullies in a forest it may be possible to describe and measure them all in some detail, even installing fixed posts for measuring soil loss and gully encroachment. At the other end of the scale, a heavily degraded, recently cleared, steeply sloping forest in the monsoon season may have abundant examples of sheet wash, rills, gullies and landslides and there may be inadequate human resources to monitor them all. In such a case, the use of photography and community sketches to analyse the rapidly changing situation is likely to be the best approach.

ISSUES AND CHALLENGES

The field methods presented in this chapter are designed to provide a simple but robust (i.e. repeatable and quantifiable) approach to the monitoring of soil erosion in forests using readily available equipment. They are intended for use by forestry officers in cooperation with local stakeholders, who can provide valuable insights to the interpretation of erosion features and their causes, effects and timescales. The description and quantification of erosion, as described here, requires initial training, which is best done predominantly in the field.

In monitoring soil erosion in forests, the following issues should be kept in mind:

- Erosion features recognized in an area and portrayed on community maps can be qualified in terms of four descriptors – type, state, extent and severity – rated according to classes. The definitions of such classes, while the best available, have a number of shortcomings that should be addressed as these methods are applied more widely. For example, classes could be tailored to better suit particular landscapes, regions and forest types.

- The simple field method for scoring observed erosion types provides a quantitative estimate for inter-site and temporal comparisons of changed erosion status. The allocation of score classes to erosion types is somewhat arbitrary, as is the procedure for summing scores when there is more than one erosion type in a study area. It is envisaged that these approaches will be improved with wider usage.
- The methods for the direct and indirect measurement of soil loss and soil accumulation have a number of potential inaccuracies. The assumptions made in undertaking such measurements should always be stated explicitly and efforts should be made to avoid potential pitfalls.
- Research is required to develop a greater understanding of the link between management activities and the type, state, extent and severity of erosion with a view to improving management to minimize the risk of erosion and to ameliorate existing erosion – particularly in areas being newly forested – to initiate from the outset improved management strategies to avoid or minimize erosion.

References

Ahrends, A., Burgess, N.D., Milledge, S.A.H., Bulling, M.T., Fisher, B., Smart, J.C.R., Clarke, G.P., Mhoro, B.E. & Lewis, S.L. 2010. Predictable waves of sequential forest degradation and biodiversity loss spreading from an African city. *Proceedings of the National Academy of Sciences* **107(**33): 14556–14561.

Aleixo, A. 1999. Effects of selective logging on a bird community in the Brazilian Atlantic forest. *Condor* 101: 537–548.

Allen, T.F.H. & Hoekstra, **T.W.** 1992. *Toward a unified ecology.* Columbia University Press, New York, USA.

Andren, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos* 71: 355–366.

Arroyo-Rodríguez, V., Aguirre, A., Benítez-Malvido, J. & Mandujano, S. 2007. Impact of rain forest fragmentation on the population size of a structurally important palm species: *Astrocaryum mexicanum* at Los Tuxtlas, Mexico. *Biological Conservation* 138: 198–206.

Asner, G.P. & Vitousek, P.M. 2005. Remote analysis of biological invasion and biogeochemical change. *Ecology* 102: 4383–4386.

Asner, G.P., Knapp, D.E. & Broadbent, E.N. 2005. Selective logging in the Brazilian Amazon. *Science* 310: 480–482.

Azevedo-Ramos, C., de Carvalho, O. & Nasi, R. 2010. Animal indicators: a tool to assess biotic integrity alter jogging tropical forests? Unpublished paper.

Bahamóndez, C., Martin, M., Müller-Using, S., Rojas,Y. & Vergara, G. 2009. *An operational approach to forest degradation. Case studies on measuring and assessing forest degradation*. Forest Resources Assessment Working Paper No. 158. FAO, Rome.

Bakker, J.D. 2008. Increasing the utility of Indicator species analysis. *Journal of Applied Ecology* 45(6): 1829–1835.

Balvanera, P., Pfisterer, A.B., Buchmann, N., He, J.S., Nakashizuka, T., Raffaelli, D. & Schmid, B. 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters* 9: 1146–1156.

Betts, M.G. & Villard, M.A. 2008**.** Landscape thresholds in species occurrence as quantitative targets in forest management: generality in space and time? *In* M.A. Villard and B.G. Jonsson, eds. *Setting conservation targets for managed forest landscapes*. Cambridge University Press, Cambridge, UK.

Bridgeland, W.T., Beier, P., Kolb, T. & Whitham, T.G. 2010. A conditional trophic cascade: birds benefit faster growing trees with strong links between predators and plants. *Ecology* 91: 73–84.

Bogaert, J., Ceulemans, R. & Salvador-Van Eysenrode, D. 2004. Decision tree algorithm for detection of spatial processes in landscape transformation. *Environmental Management* 33(1): 62–73.

CBD. 2001. Review of the status and trends of, and major threats to, the forest biological diversity. AHTEG on Forest Biological Diversity. Montreal, 12–16 November 2001. UNEP/CBD/SBSTTA/7/INF/3. Convention on Biological Diversity, Montreal, Canada.

CBD. 2005. Report of the Inter-Sessional (Second) Meeting of the AHTEG on the Review of Implementation of the Programme of Work on Forest Biological Diversity. Montreal, 28 November–2 December 2005. UNEP/CBD/SBSTTA/11/INF/2. Convention on Biological Diversity, Montreal, Canada.

Chamshama, S.A.O., Mugasha, A.G. & Zahabu, E. 2004. Stand biomass and volume estimation for miombo woodlands at Kitulangalo, Morogoro, Tanzania. *South African Forestry Journal* 200: 49–60.

Chapin, F.S., Walker, B.H., Hobbs, R.J., Hooper, D.U., Lawton, J.H., Sala, O.E. & Tilman, D. 1997. Biotic control over the functioning of ecosystems. *Science* 277: 500–504.

Chapman, C.A. 1989. Primate seed dispersal: the fate of dispersed seeds. *Biotropica* 21:148–154.

Chornesky, E.A., Bartuska, A.M., Aplet, G.H., Britton, K.O., Cummings-Carlson, J., Davis, F.W., Eskow, J., Gordon, D.R., Gottschalk, K.W., Haack, R.A., Hansen, A.J., Mack, R.N., Rahel, F.J., Shannon, M.A., Wainger, L.A. & Wigley, T.B. 2005. Science priorities for reducing the threat of invasive species to sustainable forestry. *BioScience* 55: 335–349.

Clark, D.A., Brown, S., Kicklighter, D.W., Chambers, J.Q., Thomlinson, J. R., Jian N. & Holland, E.A. 2001. Net primary production in tropical forests: an evaluation and synthesis of existing field data. *Ecological Applications* 11: 371–384.

Collinge, S.K. 1996. Ecological consequences of habitat fragmentation: implications for landscape architecture and planning. *Landscape and Urban Planning* 36: 59–77.

Davidson, C. 1998. Issues in measuring landscape fragmentation. *Wildlife Society Bulletin* 26:32–37.

Dawkins, H.C. 1964. The productivity of lowland tropical high forest and some comparisons with its competitors (tropical conifers and eucalypts). *Journal of the Oxford University Forest Society* 5(12): 15–18.

Debach, P. & Rosen, D. 1991. *Biological control by natural enemies*. Second edition. University of Cambridge Press, UK.

Diaz, S. & Cabido, M. 2001. Vive la différence: plant functional diversity matters to ecosystem processes. *Trends in Ecological Evolution* 16: 646–655.

Diaz, S., Symstad, A.J., Chapin, F.S., Wardle D.A. & Huenneke, L.F. 2003. Functional diversity revealed by removal experiments. *Trends in Ecological Evolution* 18: 140–146.

Diaz, S., Tilman, D., Fargione, J., Chapin, F.S., Dirzo, R., Kitzberger, T., Gemmill, B., Zobel, M., Vila, M., Mitchell, C., Wilby, A., Daly, G.C., Galetti, M., Laurence, W.F., Pretty, J., Naylor, R., Power, A., Harvell, D., Potts, S., Kremen, C., Griswold, T. & Eardley, C. 2005. Biodiversity regulation of ecosystem services . *In* R. Hassan, R. Scholes and N. Ash, eds. *Ecosystems and human well-being: current state and trends. Millennium ecosystem assessment Volume 1.* Island Press, Washington, DC, USA.

Diserud, O.H. & Odegaard, F. 2007. A multiple-site similarity measure. *Biology Letters* 3: 20–22.

Driscoll, D.A. & Weir, T. 2005. Beetle responses to habitat fragmentation depend on ecological traits, habitat condition, and remnant size. *Conservation Biology* 19: 182–194.

Ellison, A.M., Bank, M.S., Clinton, B.D., Colburn, E.A., Elliott, K., Ford, C.R., Foster, D.R., Kloeppel, B.D., Knoepp, J.D., Lovett, G.M., Mohan, J., Orwig, D.A., Rodenhouse, N.L., Sobczak, W.V., Stinson, K.A., Stone, J.K., Swan, C.M., Thompson, J., Von Holle, B. & Webster, J.R. 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. *Frontiers in Ecology and Environment* 3: 479–486.

Evans, K. & Guariguata, M.R. 2008. *Participatory monitoring in tropical forest management. A review of tools, concepts and lessons learned*. CIFOR, Bogor, Indonesia.

Everitt, J.H. & Deloach, C.J. 1990. Remote sensing of Chinese tamarisk (*Tamarix chinensis*) and associated vegetation. *Weed Science* 38: 273–278.

Ewers, R.M. & Didham, R.K. 2007. The effect of fragment shape and species' sensitivity to habitat edges on animal population size. *Conservation Biology* 21: 926–936.

Ewers, R.M. & Didham, R.K. 2008. Pervasive impact of large-scale edge effects on a beetle community. *Proceedings of the National Academy of Sciences* **105:** 5426–5429.

Ewers, R.M., Thorpe, S. & Didham, R.K. 2007. Synergistic interactions between edge and area effects in a heavily fragmented landscape. *Ecology* 88: 96–106.

Ewers, R.M., Kapos, V., Coomes, D.A., Lafortezza, R. & Didham, R.K. 2009. Mapping community change in modified landscapes. *Biological Conservation* 142: 2872–2880.

Fahrig, L. 2003. Effect of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution, and Systematics* 34: 487–515.

FAO. 1973. *Manual of forest inventory with special references to tropical forests*, by J.P. Lanly. Rome.

FAO. 2001. *Global ecological zoning for the global forest resources assessment 2000*. Forestry Working Paper 56. Rome.

FAO. 2002. *Proceedings: second expert meeting on harmonizing forest-related definitions for use by various stakeholders.* Rome.

FAO. 2003. *Report of the international conference on the contribution of criteria and indicators for sustainable forest management: the way forward.* Rome.

FAO. 2006a. *Global forest resources assessment 2005*. FAO Forestry Paper 147. Rome.

FAO. 2006b. *Global assessment of growing stock, biomass and carbon stock*, by L. Marklund and D. Schoene. FAO Forest Resources Assessment Programme Working Paper 106/E. Rome.

FAO. 2008. *Guidelines for country reporting to FRA 2010*. FRA Working Paper 143. Rome.

FAO. 2009. *Towards defining degradation*, by Markku Simula. FRA Working Paper 154. Rome.

FAO. 2010. *Global forest resources assessment 2010: full report.* FAO Forestry Paper 163. Rome.

FAO. 2011. *LADA local manual for local level assessment of land degradation (February 2011), sustainable land management and livelihoods: Part 1 – planning and methodological approach, analysis and reporting; Part 2 – field tools and methods*, by S. Bunning, J. McDonagh and J. Rioux, eds. Rome.

Ferreira, M.T., Aguiar, F.C. & Nogueira, C. 2005. Changes in riparian woods over space and time: Influence of environment and land use. *Forest Ecology and Management* 212: 145–159.

Fischer, J. & Lindenmayer, D. 2007. Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography* 16: 265–280.

Foley, J.A., Asner, G.P., Costa, M.H., Coe, M.T., DeFries, R., Gibbs, H.K., Howard, E.A., Olson, S., Patz, J., Ramankutty, N. & Snyder, P. 2007. Amazonia revealed: forest degradation and the loss of ecosystem goods and services in the Amazon Basin. *Frontiers in Ecology and Environment* 5: 25–32.

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L. & Holling, C.S. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review in Ecology, Evolution and Systematics* 35: 557–581.

Gathmann, A. & Tscharntke, T. 2002. Foraging ranges of solitary bees. *Journal of Animal Ecology* 71: 757–764.

Gorchov, D.L., Cornejo, F., Ascorra, C. & Jaramillo, M. 1993. The role of seed dispersal in the natural regeneration of rain forest after strip-cutting in Peruvian Amazon. *Vegetatio* 107/108: 339–349.

GPFLR. undated. Map and analyze restoration potential. Global Partnership on Forest Landscape Restoration (www.ideastransformlandscapes.org/what-we-do/map-and-analyze/).

Groeneveld, J., Alves, L.F., Bernacci, L.C., Catharino, E.L.M., Knogge, C., Metzger, J.P., Pütz, S. & Huth, A. 2009. The impact of fragmentation and density regulation on forest succession in the Atlantic rain forest. *Ecological Modelling* 220: 2450–2459.

Grogan, J., Blundell, A.G., Landis, R.M., Youatt, A., Gullison, R.E., Martinez, M., Kometter, R., Lentini, M. & Rice, R.E. 2010. Over-harvesting driven by consumer demand leads to population decline: big-leaf mahogany in South America. *Conservation Letters* 3: 12–20.

Gunderson, L. 2000. Ecological resilience: in theory and application. *Annual Review in Ecology, Evolution and Systematics* 31: 425–439.

Hanzawa, F.H., Beattie, A.J. & Culver, D.C. 1988. Directed dispersal: demographic analysis of an ant-seed mutualism. *American Naturalist* 131: 1–13.

Harris, J.A. 2003. Measurement of the soil microbial community for estimating the success of restoration. *European Journal of Soil Science* 54: 801–808.

Harris, J.A. 2009. Soil microbial communities and restoration ecology: facilitators or followers? *Science* 325: 573–574.

Hättenschwiler S., Tiunov, A.V. & Scheu, S. 2005. Biodiversity and litter decomposition in terrestrial ecosystems. *Annual Review in Ecology, Evolution and Systematics* 36: 191–218.

Hooper D.U. & Vitousek, P.M. 1997. The effects of plant composition and diversity on ecosystem processes. *Science* 277: 1302–1305.

Howe H.F. & Smallwood, J. 1982. Ecology of seed dispersal. *Annual Review in Ecology, Evolution and Systematics* 13: 201–228.

IPCC. 2003a. *Definitions and methodological options to inventory emissions from direct human-induced degradation of forests and devegetation of other vegetation types*. J. Penman, M. Gytarsky, T. Krug, D. Kruger, R. Pipatti, L. Buendia, K. Miwa, T. Ngara, K. Tanabe. and F. Wagner, eds. IPCC National Greenhouse Gas Inventories Programme, Hayama, Japan.

IPCC. 2003b. *Good practice guidance for land use, land use change and forestry*. J. Penman, M. Gytarsky, T. Krug, D. Kruger, R. Pipatti, L. Buendia, K. Miwa, T. Ngara, K. Tanabe. and F. Wagner, eds. IPCC National Greenhouse Gas Inventories Programme, Hayama, Japan.

IPCC. 2006. *IPCC guidelines for national greenhouse gas inventories.* Prepared for the Intergovernmental Panel on Climate Change by the National Greenhouse Gas Inventories Programme. Institute for Global Environmental Strategies, Tokyo, Japan.

ITTO. 2002. *ITTO guidelines for the restoration, management and rehabilitation of degraded and secondary tropical forests.* ITTO Policy Development Series No. 13. Yokohama, Japan.

Johns, A.G. 1996. Bird population persistence in Sabahan logging concessions. *Biolological Conservation* 75: 3–10.

Kagawa Y. & Maeto, K. 2009. Spatial population structure of the predatory ground beetle *Carabus yaconinus* (Coleoptera: Carabidae) in the mixed farmland-woodland Satoyama landscapes of Japan. *European Journal of Entomology* 106: 385–391.

Kalka M.B., Kunz, T.H. & Fenton, M.B. 2008. Bats limit arthropods and herbivory in a tropical forest. *Science* 320: 71–73.

Kapos, V., Lysenko, I. & Lesslie, R. 2000. *Assessing forest integrity and naturalness in relation to biodiversity*. United Nations Environment Programme-World Conservation Monitoring Centre, Cambridge, UK.

Keane, R.E., Hessburg, P.F., Landres, P.B. & Swanson F.J. 2009. The use of historical range and variability (HRV) in landscape management. *Forest Ecology and Management* 258: 1025–1037.

Kenis, M., Auger-Rozenberg, M. A., Roques, A., Timms, L., Péré, C., Cock, M. J. W., Settele, J., Augustin, S. & Lopez-Vaamonde, C. 2009. Ecological effects of alien invasive insects. *Biological Invasions* 11: 21–45.

Klein, A-M, Vaissiere, B.E., Cane, J.H., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C. & Tscharntke, T. 2007. Importance of pollinators in changing landscapes for world crops. *Proceedings of the Royal Society B* 274: 303–313.

Kleine, M., Shaharuddin, G. & Kant, P. 2009. *Addressing forest degradation in the context of joint forest management in Udaipur, India. Case studies on measuring and assessing forest degradation.* FRA Working Paper No. 157. FAO, Rome.

Kometter, R.F., Martinez, M., Blundell, A.G., Gullison, R.E., Steininger, M.K. & Rice, R.E. 2004. Impacts of unsustainable mahogany logging in Bolivia and Peru. *Ecology and Society* 9(1): Art. 12 (http://www.ecologyandsociety.org/vol9/iss1/art12).

Kupfer, J.A., Malanson, G.P. & Franklin, S.B. 2006. Not seeing the ocean for the islands: the mediating influence of matrix-based processes on forest fragmentation effects. *Global Ecology and Biogeography* 15: 8–20.

Lambin, E.F. 1999. Monitoring forest degradation in tropical regions by remote sensing: some methodological issues. *Global Ecology and Biogeography* 8(3-4): 191–198.

Laurance, W.F., Lovejoy, T.E., Vasconcelos, H.L., Bruna, E.M., Didham, R.K., Stouffer, P.C., Gascon, C., Bierregaard, R.O., Laurance, S.G. & Sampaio, E. 2002. Ecosystem decay of Amazonian forest fragments: a 22-year investigation. *Conservation Biology* 16: 605–618.

Lawrence, A., Kinhal, G., Luintel, H., Molteno, S. & Gillett, S. 2008. *Participatory science for sustainable wild harvests: a method book*. University of Oxford, Oxford, UK.

Lawton, J.H., Bignell, D.E., Bolton, B., Bloemers, G.F., Eggleton, P., Hammond, P.M., Hodda, M., Holt, R.D., Larsen, T.B., Mawdsley, N.A., Stork, N.E., Srivastava, D.S. & Watt, A.D. 1998. Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. *Nature* 391: 72–76.

Lennon, J.J., Koleff, P., Greenwood, J.J.D. & Gaston, K.J. 2001. The geographical structure of British bird distributions: diversity, spatial turnover and scale. *Journal of Animal Ecology* **70**: 966–979.

Lewandowski, A.S., Noss, R.F. & Parsons, D.R. 2010. The effectiveness of surrogate taxa for the representation of biodiversity. *Conservation Biology* 24: 1367–1377.

Leys, J., Butler, H., Yang, X. & Heidenreich, S. 2010. *CEMSYS modelled wind erosion. PRN 0808 – 1220*. Department of Environment, Climate Change and Water, Sydney, Australia. http://www.nrm.gov.au/publications/books/pubs/cemsys-wind-erosion.pdf

Lindenmayer, D. & Fischer, J. 2007. Tackling the habitat fragmentation panchreston. *Trends in Ecology and Evolution* 22(3): 127–132.

Lindenmayer, D.B., Manning, A.D., Smith, P.L., Possingham, H.P., Fischer, J., Oliver, I. & McCarthy, M.A. 2002. The focal-species approach and landscape restoration: a critique. *Conservation Biology* 16: 338–345.

Logan, J.A. & Powell, J.A. 2001. Ghost forests, global warming, and the mountain pine beetle. *American Entomology* 47(3): 160–172.

Lucier, A., Ayers, M., Karnosky, D. & Thompson, I. 2009. Forest responses and vulnerabilties to recent climate change. *In* R. Seppala, A. Buck and P. Katila, eds. *Adaptation of forests and people to climate change: a global assessment report.* IUFRO World Series Volume 22. Vienna, Austria.

Lund, H.G. 2009. *What is a degraded forest?* Forest Information Services, Gainesville, USA.

Maleque, M.D.A., Maeto, K., Makino, S., Goto, H., Tanaka, H., Hasegawa, M. & Miyamoto, S. 2010. A chronosequence of understorey parasitic wasp assemblages in secondary broad-leaved forests in a Japanese 'satoyama' landscape. *Insect Conservation and Diversity* 3: 143–151.

Mason, D. 1996. Responses of Venezuelan understory birds to selective logging, enrichment strips, and vine cutting. *Biotropica* 28: 296–309.

McGarigal, K., Cushman, S.A., Neel, M.C. & Ene, E. 2002. FRAGSTATS: spatial pattern analysis program for categorical maps. Computer software program (www.umass.edu/landeco/research/fragstats/fragstats.html).

Meyer, O. 1994. Functional groups of microorganism. *In* E.D. Schulze and H.A. Mooney, eds. *Biodiversity and ecosystem function.* Springer, New York, USA.

Millennium Ecosystem Assessment. 2005. *Ecosystems and human well-being: biodiversity synthesis.* World Resources Institute, Washington, DC, USA.

Mulder, P. & McGarry, D. 2010. Soil erosion indicators. Internal report of the project: 'Rural development project: contract 3 – sustainable pasture, arable and forest land management – Tajikistan'. GITEC/DMC/Asian Development Bank/GEF. Tajikistan.

Niemi, G.J., Hanowski, J.M., Lima, A.R., Nicholls, T. & Weiland, N. 1997. A critical analysis on the use of indicator species in management. *Journal of Wildlife Management* 61(4): 1240–1252.

Nieuwenhuis, M. 2000. *Terminology of forest management.* IUFRO World Series Volume 9. IUFRO 4.04.07, International Union of Forestry Research Organizations, Vienna, Austria.

Noss, R.F. 1999. Assessing and monitoring forest biodiversity: a suggested framework and indicators. *Forest Ecology and Management* 115: 135–146.

Nummelin, M. & Fursch, H. 1992. Coccinelids of the Kibale forest, western Uganda: a comparison between virgin forest and managed sites. *Tropical Zoology* 5: 155–166.

Oliver, I. & Beattie, A.J. 1996. Designing a cost-effective invertebrate survey: a test of methods for rapid assessment of biodiversity. *Ecological Applications* 6: 594–607.

Oliver, I., Nally, R.M. & York, A. 2000. Identifying performance indicators of the effects of forest management on ground-active arthropod biodiversity using hierarchical partitioning and partial canonical correspondence analysis. *Forest Ecology and Management* 139: 21–40.

Oren R., Ellsworth, D.S., Johnsen, K.H., Phillips, N., Ewers, B.E., Maier, C., Schafer, K.V.R., McCarthy, H., Hendrey, G., McNulty, S.G. & Katul, G.G. 2001. Soil fertility limits carbon sequestration by forest ecosystems in a CO_2-enriched atmosphere. *Nature* 411(6836): 469–472.

Pauchard, A. & Maheu-Giroux, M. 2007. *Acacia dealbata* invasion across multiple scales: Conspicuous flowering species can help us study invasion pattern and processes. *In* H. Strand, R. Höft, J. Strittholt, L. Miles, N. Horning, E. Fosnight and W. Turner, eds. *Sourcebook on remote sensing and biodiversity indicators.* Technical Series No. 32. Secretariat of the Convention on Biological Diversity, Montreal, Canada.

Peres, C.A., Baider, C., Zuidema, P.A. Wadt, L.H.O., Kainer, K.A., Gomes-Silva, D.A.P., Salomão, R.P., Simões, L.L., Franciosi, E.R.N., Valverde, F.N., Gribel, R., Shepard Jr., G.H., Kanashiro, M., Coventry, P., Yu, D.W., Watkinson, A.R. & Freckleton, R.P. 2003. Demographic threats to the sustainability of Brazil nut exploitation. *Science* 302: 2112–2114.

Pimental, D., Zuniga, R. & Morrison, D. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics* 52: 273–288.

Potvin, C., Mancilla, L., Buchmann, N., Monteza, J., Moore, T., Murphy, M., Oelmann, Y., Scherer-Lorenzen, M., Turner, B.L., Wilcke, W., Zeugin, F. & Wolf, S. 2011. An ecosystem approach to biodiversity effects: carbon pools in a tropical tree plantation. *Forest Ecology and Management* 261: 1614–1624.

Ramsey, E.W., Nelson, G.A., Sapkota, S.K., Seeger, E.B. & Martella, K.D. 2002. Mapping Chinese tallow with color infrared photography. *Photogrammetric Engineering and Remote Sensing* 68: 251–255.

Ricketts T.H., Daily, G.C., Ehrlich, P.R. & Michener, C.D. 2004. Economic value of tropical forest to coffee production. *Proceedings of the National Academy of Sciences* 101: 12579–12582.

Ritz, K., Black, H.I.J., Campbell, C.D., Harris, J.A. & Wood, C. 2009. Selecting biological indicators for monitoring soils: a framework for balancing scientific and technical opinion to assist policy development. *Ecological Indicators* 9: 1212–1221.

Rolstad, J., Gjerde, I., Gundersen, V.S. & Saetersdal, M. 2002. Use of indicator species to assess forest continuity: a critique. *Conservation Biology* 16: 253–257.

Russell, A.E, Raich, J.W., Arrieta, R.B., Valverde-Barrantes, O. & González, E. 2010. Impact of individual tree species on carbon dynamics in a moist tropical forest environment. *Ecological Applications* 20: 1087–1100.

Saura, S. & Carballal, P. 2004. Discrimination of native and exotic forest patterns through shape irregularity indices: an analysis in the landscapes of Galicia, Spain. *Landscape Ecology* 19: 647–662.

Schulze, M., Grogan, J., Uhl, C., Lentini, M. & Vidal, E. 2008. Evaluating *ipê* (*Tabebuia*, Bignoniaceae) logging in Amazonia: sustainable management or catalyst for forest degradation? *Biological Conservation* 141: 2071–2085.

Secretariat of the Convention on Biological Diversity. 2010. *Global biodiversity outlook 3*. Montréal, Canada.

Shackleton, S., Shanley P. & Ndoye, O. 2007. Invisible but viable: recognizing local markets in non-timber forest products. *International Forestry Review* 9(3): 697–712.

Sheil, D., Nasi, R. & Johnson, B. 2004. Ecological criteria and indicators for tropical forest landscapes: challenges in the search for progress. *Ecology and Society* 9(1): art. 7 (http://www.ecologyandsociety.org/vol9/iss1/art7).

Souza, Jr. C., Firestone, L., Silva, L.M. & Roberts, D. 2003. Mapping forest degradation in the Eastern Amazon from SPOT4 through spectral mixture models. *Remote Sensing of Environment* 87: 494–506.

Stockdale, M. 2005. Steps to sustainable and community-based NTFP management. NTFP Exchange Programme for South and South East Asia, Philippines.

Stocking, M.A. & Murnaghan, N. 2001. *Handbook for the field assessment of land degradation*. Earthscan Publications, London, UK.

Strand, H., Höft, R., Strittholt, J., Miles, L., Horning, N., Fosnight, E. & Turner, W., eds. 2007. *Sourcebook on remote sensing and biodiversity indicators.* Technical Series No. 32. Secretariat of the Convention on Biological Diversity, Montreal, Canada.

Strassburg, B.B.N., Kelly, A., Balmford, A., Davies, R.G., Gibbs, H.K., Lovett, A., Miles, L., Orme, C.D.L., Price, J., Turner, R.K. & Rodrigues, A.S.L. 2010. Global congruence of carbon storage and biodiversity in terrestrial ecosystems. *Conservation Letters* 3: 98–105.

Straube, D., Johnson, E.A., Parkinson, D., Scheu, S. & Eisenhauer, N. 2009. Nonlinearity of effects of invasive ecosystem engineers on abiotic soil properties and soil biota. *Oikos* 118: 885–896.

Taki, H., Okabe, K., Yamaura, Y., Matsuura, T., Sueyoshi, M., Makino, S. & Maeto, K. 2010. Effects of landscape metrics on *Apis* and non-*Apis* pollinators and seed set in a self-incompatible crop. *Basic and Applied Ecology* 11: 594–602.

Theron, J.M., Laar, A. Van, Kunneke, A. & Bredenkamp, B.V. 2004. A preliminary assessment of utilizable biomass in invading *Acacia* stands on the Cape coastal plains. *South African Journal of Science* 100: 123–125.

Thompson, I.D. & Welsh, D.A. 1993. Integrated resource management in boreal forest ecosystems – impediments and solutions. *Forestry Chronicle* 69: 32–39.

Thompson, I., Mackey, B., McNulty, S. & Mosseler, A. 2009. *Forest resilience, biodiversity, and climate change. A synthesis of the biodiversity/resilience/stability relationship in forest ecosystems.* Technical Series No. 43. Secretariat of the Convention on Biological Diversity, Montreal, Canada.

Top, N., Mizoue, N. & Kai, S. 2004. Estimating forest biomass increment based on permanent sample plots in relation to woodfuel consumption: a case study in Kampong Thom Province, Cambodia. *Journal of Forestry Research* 9: 117–123.

Top, N., Mizoue, N., Ito, S., Kai, S., Nakao, T. & Ty, S. 2006. Re-assessment of woodfuel supply and demand relationships in Kampong Thom Province, Cambodia. *Biomass and Bioenergy* 30(2): 134–143.

Tsai, F., Lin, E. & Wang, H. 2005. Detecting invasive plant species using hyperspectral satellite imagery. GEOScience and Remote Sensing Symposium, 2005, IGARSS '05 Proceedings. IEEE International 4: 3002–3005.

UNEP-WCMC. 2007. Global distribution of current forests (http://www.unep-wcmc.org/forest/global_map.htm, accessed 2007).

United Nations. 2007. Non-Legally Binding Instrument on All Types of Forests. United Nations General Assembly. A/C.2/62/L.5. 22 October 2007.

Van der Meer, F., Schmidt, K.S., Bakker, A. & Bijker, W. 2002. New environmental remote sensing systems. In A.K. Skidmore, ed. *Environmental modelling with GIS and remote sensing*. Taylor and Francis, London, UK.

Vasconcelos, H.L., Vilhena, J.M.S. & Caliri, G.J.A. 2000. Responses of ants to selective logging of a central Amazonian forest. *Journal of Applied Ecology* 37: 1–8.

Wadt, L.H.O., Kainer, K.A., Staudhammer, C.L. & Serrano, R.O.P. 2008. Sustainable forest use in Brazilian extractive reserves: natural regeneration of Brazil nut in exploited populations. *Biological Conservation* 141: 332–346.

Walker, B.H. 1992. Biological diversity and ecological redundancy. *Conservation Biology* 6: 18–23.

Walker, B.H., Holling, C.S., Carpenter, S.R. & Kinzig, A.P. 2004. Resilience, adaptability and transformability in socio-ecological systems. *Ecology and Society* 9(2): art. 5 (http:www.ecology and society.org/vol9/iss2/art5).

Watling, J.I. & Donnelly, M.A. 2006. Fragments as islands: a synthesis of faunal responses to habitat patchiness. *Conservation Biology* 20: 1016–1025.

Westphal, C., Bommarco, R., Carré, Lamborn, E., Morison, N., Petanidou, T., Potts, S.G., Roberts, S.P.M., Szentgyorgy, H., Tscheuline, Vaissiè, B.E., Woyciechowski, M., Biesmeijer, J.C., Kunin, W.E., Settele, J. & Steffan-Dewenter, I. 2008. Measuring bee diversity in different European habitats and biogeographical regions. *Ecological Monographs* 78: 653–671.

Whitaker, R.H. 1972. Evolution and measurement of species diversity. *Taxon* 2: 213–251.

Whitmore, T.C. 1998. *An Introduction to tropical rain forests.* Second edition. Oxford University Press, New York, USA.

Wu, J., Huang, J., Han, X., Xie, Z. & Gao, X. 2003. Three-Gorges Dam: experiment in habitat fragmentation? *Science* 300: 1239–1240.

Yamaura, Y., Ikeno, S., Sano, M., Okabe, K. & Ozaki, K. 2009. Bird responses to broad-leaved forest patch area in a plantation landscape across seasons. *Biological Conservation* 142: 2155–2165.

Yeats, G.W. 2003. Nematodes as soil indicators: functional and biodiversity aspects. *Biology and Fertility of Soil* 37: 199–210.

Yoshimura, M. 2007. Comparison of stream benthic invertebrate assemblages among forest types in the temperate region of Japan. *Biodiversity Conservation* 16: 2137–2148.

Zhang, Y., Duan, B., Xian, J., Korpelainen, H. & Li, C. 2011. Links between plant diversity, carbon stocks, and environmental factors along a successional gradient in a subalpine coniferous forest in southwest China. *Forest Ecology and Management* 262: 361–369.

Ziegler, A.D., Giambelluca, T.W., Plondke, D., Leisz, S., Tran, L.T., Fox, J., Nullet, M.A., Vogler, J.B., Minh Troung, D. & Tran, D.V. 2007. Hydrological consequences of landscape fragmentation in mountainous northern Vietnam: buffering of Hortonian overland flow. *Journal of Hydrology* 337: 52–67.

ANNEX 1
Preparatory background information

SURVEY OF FOREST DEGRADATION

As part of the Global Forest Resources Assessment 2010 (FAO 2010), questionnaires on forest degradation were sent to 177 national correspondents, and responses were received from correspondents in 45 countries.

One-third of responding countries had developed a national definition of forest degradation. Typical indicators in definitions included stocking level, productivity, biomass density and species composition, while the most common reference level appeared to be 'what is expected on the site'. Some countries have assessed degradation without developing an explicit definition. Half of responding countries had definitions for related terms.

Two-thirds of responding countries did not determine degradation according to different purposes of management and did not consider human-induced temporary changes as degradation. Most had no assessment methodology and fewer than half provided an actual or estimated figure of degradation.

The analysis suggests that the majority of countries consider that the elements of SFM provide a potential framework for the analysis of forest degradation.

Causes of forest degradation were listed primarily by respondents as illegal logging, fire, fuelwood collection and shifting cultivation. In Asia, pests, diseases and insect infestations were also listed. In Sahelian countries, grazing, drought and fuelwood collection were listed as key factors. In Europe, fire, pests, diseases and wind damage were listed, and Pacific Island states listed wind, coastal erosion, fuelwood collection, development projects, pests and diseases.

Analytical study of definitions

An analysis (FAO 2009) was conducted of existing international and national definitions of forest degradation and their elements, parameters and commonalities and differences. It was determined that the generic definition of forest degradation – the reduction of the capacity of a forest to provide goods and services – provides a common framework for all international definitions and is compatible with an ecosystem-services approach. The most comprehensive international definitions have been developed by ITTO and the CBD, covering change in forest structure and dynamics, forest functions, human induced causes and a reference state. In these definitions the spatial scale is at the stand or site level and the temporal scale is usually long-term. The definition used by FAO's 2000 Global Forest Resources Assessment covers many similar elements but does not specifically address the causes of deforestation. The definition developed by the Intergovernmental Panel on Climate Change focuses on human-induced changes to the carbon cycle.

Case studies

Twenty case studies were assembled from diverse contexts worldwide. Assessment methodologies were generally scarce compared with information on the causes, drivers and effects of forest degradation. Some elements of SFM, as they relate to forest degradation, were studied more than others.

In several case studies the best approach to monitoring, assessing and reporting on forest degradation was considered to be to combine the use of satellite imagery with supportive ground-based inventory.

Community-based approaches, particularly where government and communities jointly manage forests, appear to also be quite effective at both obtaining information and improving management. Community-based case studies included approaches in Ghana, India and Niger, and a 'community carbon' project.

Some of the more promising methods identified for monitoring and assessing forest degradation included:

- a combination of remote sensing, GIS and field observations;
- the use of advanced technologies such as aerial laser scanning;
- community-based assessment.

Technical meeting

A technical meeting on the assessment and monitoring of forest degradation was convened at FAO headquarters in Rome, Italy in September 2009, attended by 37 specialists from 15 countries and 12 international forest-related organizations and processes. Participants reviewed the definitions analysis and the case studies and discussed ways to improve the measurement, assessment and reporting of forest degradation.

The main conclusions of the meeting were that:

- the generic definition of 'forest degradation' – a reduction in the capacity of a forest to provide goods and services – was a suitable starting point for approaching the issue;
- the many aspects of forest degradation should be better communicated to parties to and relevant stakeholders of forest-related international conventions;
- attention should be focused on harmonizing definitions and methods for monitoring five aspects of forest degradation – stocking level, biological diversity, forest health, forest goods obtained compared with sustainably managed forests, and forest soils;
- methodologies exist for monitoring change in carbon stocks and therefore it is possible to include forest degradation in the proposed REDD mechanism.

There was a call for the development of tools and guidelines for measuring the various aspects of forest degradation. Presentations made at the meeting are available at www.fao.org/forestry/cpf/degradation/en/.

REFERENCES

FAO. 2009. *Towards defining degradation*, by Markku Simula. FRA Working Paper 154. Rome.

FAO. 2010. *Global forest resources assessment 2010: full report.* FAO Forestry Paper 163. Rome.

ANNEX 2

Examples of equations for volume, biomass and carbon estimations

VOLUME

Stem volume (growing stock or dead wood). Example of a volume function for a tree stem, including bark:

$$volume_{stem} = (dbh^2)/4 \times h_{tot} \times \pi \times f_{form}$$

(cylinder volume adjusted with a stem form factor)

Where

$volume_{stem}$	= *tree stem volume, including bark (= growing stock if including only living trees; = dead wood if including only dead/dying trees)*
dbh	= *tree stem diameter at breast height*
$htot$	= *tree total height/length*
π	= *3.141596*
f_{form}	= *tree stem form factor (usually in the range 0.3–0.8; ~0.5 for broadleaved trees species; ~0.65 for coniferous tree species).*

Commercial stem volume. Example of commercial volume function of a tree stem including bark:

$$volume_{comm} = (dbh^2)/4 \times h_{comm} \times \pi \times f_{comm}$$

(cylinder volume adjusted with a stem form factor)

Where

$volume_{comm.}$	= *commercial tree stem volume, including bark (for commercial tree species)*
dbh	= *tree stem diameter at breast height*
h_{comm}	= *commercial tree stem height/length (according to national definitions)*
f_{comm}	= *tree commercial stem form factor (generally in the range 0.5–0.9; ~0.7 for broadleaved trees species; ~0.8 for coniferous tree species).*

Branch volume (for timber estimates). Example of volume function of a branch, excluding bark, to estimate timber volume:

$$volume_{branch} = (d_{avg}^2)/4 \times l_{timber} \times \pi$$

Where

$volume_{branch}$	= *branch volume, excluding bark (= branch timber volume)*
d_{avg}	= *average branch diameter (or diameter at the middle of the timber length) under bark*
l_{timber}	= *branch timber length (to where the branch reaches minimum diameter suitable for timber production).*

Harvested stem volume. Example of volume function of a harvested and extracted tree stem, including bark, and an optional reduction for stump volume if stump is above the default stump height:

$$volume_{extract} = (dsh \times f_{dsh})^2)/4 \times H_{table} \times \pi \times f_{form} [- (dsh^2)/4 \times (H_{stump} - H_{def}) \times \pi \times f_{sred}]$$

(cylinder volume based on an adjusted stump diameter, to approximate dbh, and a tree stem form factor)

Where

$volume_{extract}$ = *extracted tree stem volume, including bark*

d_{sh} = *stump diameter (at stump height or at 1.3m if higher than dbh)*

f_{dsh} = *stump diameter adjustment factor (usually in the range 0.6–1.0) to approximate dbh of extracted tree stem*

h_{table} = *tree total height/length (from height–diameter table, which can be generated using forest inventory data)*

f_{form} = *tree stem form factor (usually in the range 0.3–0.8; ~0.5 for broadleaved trees species and ~0.65 for coniferous tree species)*

h_{stump} = *stump height*

h_{def} = *stump default height – the height at which tree is usually felled (usually in the range 0.2–0.5 m)*

f_{sred} = *stump reduction form factor (usually in the range 0.8–1.2) – option factor to approximate the length of stem not extracted if stump height is above the default stump height.*

Stump volume (to estimate extractable stump volume). Example of equation for stump volume, including bark and top coarse roots:

$$volume_{stump} = (dsh^2)/4 \times h_{stump} \times \pi \times f_{stump}$$

(cylinder volume adjusted with a stump form factor)

Where

d_{sh} = *stump diameter (at stump height or at 1.3 m if higher than dbh)*

h_{stump} = *stump height*

f_{stump} = *stump form factor (usually in the range 1.3–2.0).*

BIOMASS

Stem biomass (growing-stock biomass, dead-wood biomass). Examples of equations for stem biomass, growing stock biomass and dead wood biomass:

$$b_{stem} = volume_{stem} \times WD$$
$$b_{gs} = volume_{gs} \times WD_{gs}$$
$$b_{dw} = volume_{dw} \times WD_{dw}$$

Where

b_{stem} = *stem biomass (dry biomass of stem, including bark)*

b_{gs} = *growing-stock biomass (dry biomass of stem, including bark, for living trees)*

b_{dw} = *dead wood biomass (dry biomass of stem, including bark, for dead/dying trees)*

$volume_{stem}$ = *tree stem volume, including bark and including both living trees, and dead/dying trees (usually merchantable or bole height volume)*

$volume_{gs}$ = *tree stem volume, including bark and including only living trees*
$volume_{dw}$ = *tree stem volume, including bark and including only dead/dying trees*
WD = *wood density (tonnes dry matter per* m^3 *of stem volume; see Table 3A.1.9-1 and Table 3A.1.9-1 in IPCC 2006 for default factors)*
WD_{gs} = *growing stock wood density (tonnes dry matter per* m^3 *of stem volume).*
WD_{dw} = *dead wood density (tonnes dry matter per* m^3 *of stem volume).*

Above-ground biomass. Example of an equation for stem biomass:

$$b_{a\text{-}g} = b_{gs} \times BEF$$

Where
$b_{a\text{-}g}$ = *total above-ground living biomass (stem, bark, branches, leaves, fruits, flowers, nuts, etc., in living tree)*
b_{gs} = *growing-stock biomass (dry biomass of stem, including bark, for living trees)*
BEF = *biomass expansion factor (expanding growing-stock biomass to total above-ground biomass; see Table 3A.1.10 in IPCC 2006 for default factors).*

Below-ground biomass. Example of an equation for below-ground biomass:

$$b_{b\text{-}g} = b_{a\text{-}g} \times R_{r\text{-}s}$$

Where
$b_{b\text{-}g}$ = *below-ground living biomass (i.e. living tree roots)*
$b_{a\text{-}g}$ = *total above-ground living biomass (i.e. stem, bark, branches, leaves, fruits, flowers, nuts, etc. of living tree)*
$R_{r\text{-}s}$ = *root–shoot ratio of below-ground and above-ground biomass (usually expressed in tonnes of dry matter) (see Table 3A.1.8 in IPCC 2006 for default factors).*

.

ANNEX 3
Additional descriptors of erosion features[14]

GULLY EROSION

A gully is a deep depression, channel or ravine in a landscape, looking like a recent and very active extension to natural drainage channels. Gullies may be continuous or discontinuous. A continuous gully occurs where the bed of the gully is at a lower angle slope than the overall land slope. Discontinuous gullies erode at the upslope head but deposit sediment at the lower end of the discontinuity. Hence, several discontinuous gullies may occupy the same landscape depression, their shapes progressively moving upslope. Gullies are obvious features in a landscape. They may be very large (metres wide and deep) and can undermine buildings, roads and trees.

A gully is caused by the action of water. Runoff is channelled into grooves that deepen over time to form distinct heads with steep sides. Gullies extend and deepen in an up-valley direction by waterfall erosion and the progressive collapse of their upslope parts. Gully sides may collapse as a result of water seepage or undermining by water flow within the gully.

Several conditions are conducive to gully development. Gullies tend to form where slopes are long and land use has resulted in a loss of vegetation and the exposure of the soil surface over a large area, so that there is more runoff. Gullies are particularly prevalent in deep loamy to clayey materials, in unstable clays (e.g. sodic soils), on pediments immediately downslope of bare rock surfaces, and on very steep slopes subject to water seepage and to landslides.

Mass movement

Mass movement is the relatively large downslope movement of soil and/or rock (landslides, slumps, earth flows and debris avalanches), which can be caused by water or earthquakes.

Mudflows

Mudflows occur when unconsolidated materials become saturated with water during snowmelt or rainstorms and flow downslope.

Rill erosion

Rills are caused by the scouring action of water as it runs downslope during rain events, creating shallow linear channels in the soil surface that deepen over time. A broadly accepted distinction between rills and gullies, often applied in soil conservation, is that the former can be eliminated using normal agronomic practices (such as ploughing), whereas gullies require specific large interventions such as bulldozers, concrete lining or gabions (rock-filled bolsters placed in a gully to accumulate sediment). Rills tend to occur on slopes and gullies along drainage lines.

Rills occur on sloping surfaces where runoff is high and where soil has been disturbed but the surface is left relatively smooth and unvegetated (e.g. after forest clearing, tillage and building construction and on the sides of earthen dams and road embankments). Rills are also

[14] Based on information contained in an early draft of FAO (2011).

likely to form in slight depressions in the soil; thus, rills may develop in paths, roadways and culverts and in the tracks made by timber extraction vehicles and tillage equipment.

Streambank erosion

Linear erosion can occur on the banks, floodplains and alluvial deposition zones of streams and rivers. Severe streambank erosion can affect the quality and quantity of water supplies, damage infrastructure and pose risks to people and their livelihoods.

Rotational slumping

Rotational slumps are landslides that occur when a slumping block slides on a curved failure surface, causing the upper surface to tilt back.

Sheet wash

Sheet-wash erosion is caused by surface runoff that spreads across a rainfall event (i.e. when the soil infiltration rate has been exceeded), picking up and transporting soil particles dislodged by the impact of raindrops. Sheet-wash erosion is a gradual, uniform process that is difficult to detect until it develops into rill erosion.

Rain-splash erosion

Rain-splash erosion occurs when raindrops displace soil particles vertically and downslope. It may also create a compacted surface crust that inhibits tree seedling establishment.

Terracettes

Terracettes are small horizontal ridges that form irregular, step-like formations on moderate to steep hillslopes. In some cases terracettes form as a result of the expansion of soil particles in saturated conditions and their subsequent contraction as they dry, which causes them to move slowly downhill, a process known as soil creep. Terracettes are common on hillsides used for pasture (especially sheep and goats), where soil creep is greatly exacerbated by trampling by livestock.

Tunnelling[15]

Tunnelling is an insidious form of subsurface erosion and can cause considerable damage even before it manifests on the surface. Tunnel erosion is caused by the movement of excess water through a dispersive (usually sodic) subsoil.

Compacted bare areas generate runoff that flows directly into the subsoil via surface cracks, animal burrows or old root holes. Once concentrated in the subsoil the water causes sodic clays to disperse and form a suspension or slurry. Provided there is sufficient gradient, the slurry will flow downslope beneath the soil surface, re-emerging where the subsoil is exposed (e.g. by erosion or construction work). Once formed, a tunnel will continue to enlarge during subsequent wet periods to the point where parts of the roof collapse, forming potholes and erosion gullies.

Large tunnels and those that have already begun to collapse and form gullies are easily identified. Less conspicuous are the smaller potholes and outlet holes associated with newly formed or forming tunnel erosion.

Wind erosion

Wind erosion is soil degradation by the action of the wind which abrades, transports and deposits particles of soil and sand. The extent and severity of wind erosion depend on the soil type, climate, vegetation cover and the speed and frequency of the wind.

[15] For more information see www.landcare.net.au/files/fieldguidebook/tunero.html.

With wind erosion there can be either one or both of:

- Deflation – the action of removal of soil/sand particles resulting in a loss of the surface soil layer, the appearance of a stony surface and the exposure of plant roots. Deflation is sometimes accompanied by corrosion.
- Accumulation – the deposition of wind-transported soil/sand particles when the wind loses speed or becomes too laden. It can take one of the following forms according to severity
  - severely affected areas may form well-developed dune fields with or without vegetation
  - in moderately affected areas there may be an accumulation of material trapped at the edges of fields or along roads
  - in weakly affected areas there may be diffuse accumulations in the form of sandy layers around herbaceous vegetation and fine sand deposits less than 2–3 cm deep.

ANNEX 4

Illustrative examples of erosion features[16]

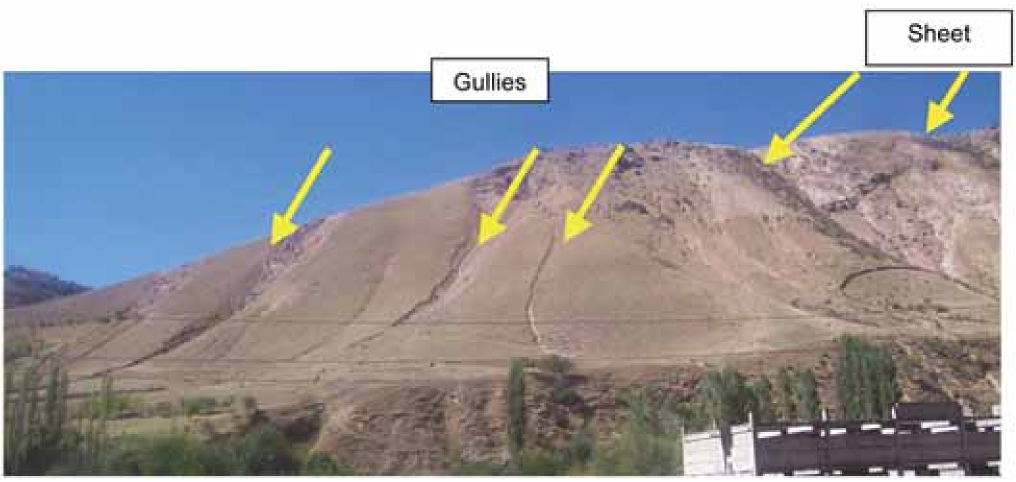
Sheet wash

[16] Unless otherwise noted, photos courtesy D. McGarry in Tajikistan (Mulder and McGarry, 2010).

Rill erosion

Source (lower photo): United States Department of Agriculture Natural Resources Conservation Service, Kansas, United States (http://www.google.com.au/imgres?imgurl=http://www.ks.nrcs.usda.gov/).

Gully



Ravine

Landslide

Rotational slumping

Terracettes

Tunnelling

Source: Department of Primary Industries, Victoria, Australia (http://www.dse.vic.gov.au/dpi/vro/vroimages.nsf/Images/tunnel_erosion_caption/$File/tn_turbid_tunnelflow.jpg).

Streambank erosion

ANNEX 5
Soil erosion field worksheets

FIELD WORKSHEET #1
Soil erosion in forests

Part A: Soil erosion observations

A. 1: Community map
Draw all visible erosion features and note or represent other relevant information (e.g. vegetation, main land uses, slopes, villages, roads and streams), and the location (latitude, longitude, elevation and north point) of the observation point.

A.2: Site details
Date:
Author(s):
Site location (latitude & longitude):
Elevation (m):
Slope(s) in degrees:
Forest type (current and past):
Forest age:
Recent weather conditions:

FIELD WORKSHEET #2
Soil erosion in forests

Part A: Soil erosion observations

A. 2: Recording and (scoring) of type, state, extent and severity
– of observed erosion features in a study area

	Type (score)	State (score)	Extent (score)	Severity (score)	TOTAL SCORE
Type #example	gully (4)	partly stabilised (2)	moderate (2)	extreme (4)	
Type 1					
Type 2					
Type 3					
Type 4					
				Sum (total) of all scores of all types	

Part B: Field measurements of soil erosion features

B. 1: Field measurement of soil erosion features
– to gain more quantified data on rates of soil erosion

Data required for individual (i) direct and (ii) indirect features

Erosion feature	***Measurements required***		
Rill	Width =	Depth =	Catchment area =
Gullies / ravines	Width (lip) =	Width (base) =	Depth = Length =
1. Plant/tree root exposure **2. Fence posts** **3. Tree mounds** **4. Rock notches & coloration**	Difference in height - current soil surface and 1. Top tree root = 2. Original soil level = 3. Soil not under tree = 4. Current soil level =		
Pedestals	Difference height - current soil surface and bottom of stone =		
Armour layer	Depth of coarse layer =	Coarse fragments as % of total in layer =	
Soil against barrier	Soil depth at barrier = Length of barrier =	Accumulation length upslope =	Catchment area =

I2479E/1/11.11