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CRITERIA AND INDICATORS FOR SUSTAINABLE FOREST MANAGEMENT

EDITED BY R.J. RAISON, A.G. BROWN AND D.W. FLINN

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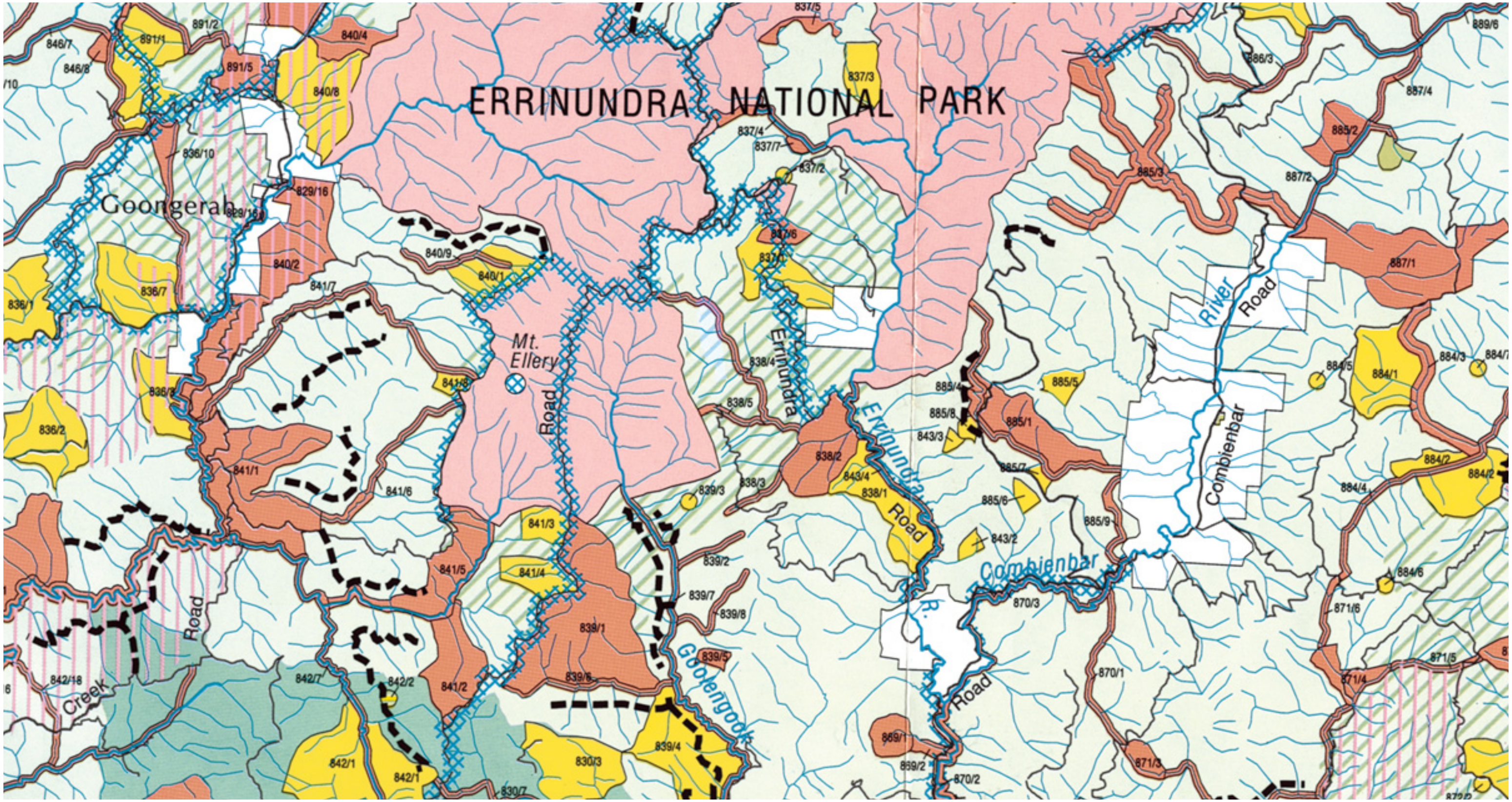
A mature wet sclerophyll eucalypt forest in East Gippsland, Victoria. The dominant tree species are *Eucalyptus obliqua* and *Eucalyptus cypellocarpa*, with an understorey of wattle (*Acacia dealbata*) and treeferns (*Cyathea australis*). These forests have both high conservation and wood production values. Within a landscape context the forest is zoned into areas managed for conservation and areas that are managed for multiple-use including wood production. Indicators are required to guide management for a broad range of values in such forests.



Map showing zoning of native forest use in East Gippsland, Victoria summarizing the outcomes of a comprehensive planning process. The zones reflect varying management emphasis ranging from complete conservation (shaded pink), special protection (brown), special management to protect threatened species and cultural values (yellow), general management for multiple use (white) and intensive wood production (green). Extensive stakeholder input occurred during the development of management objectives for each zone and the specification of their areal extent. Ideally, indicators and targets should be specified in forest management plans that can be used to assess progress against agreed objectives.

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Criteria and Indicators for Sustainable Forest Management

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In December 1996, the Executive Board of the International Union of Forestry Research Organizations (IUFRO) established a Task Force on Sustainable Forestry to provide strategic advice to the Board, to stimulate research on appropriate criteria and indicators to support sustainable forest management, and to prepare a state-of-knowledge report for the IUFRO World Congress in 2000. One of the first initiatives of this Task Force was to organize a series of international meetings on sustainable forestry. The first of these was a conference on Criteria and Indicators for Sustainable Forest Management held in Melbourne, Australia in August 1998. As described in Chapter 1, that meeting established the foundations for this book.

The Melbourne Conference was hosted by the Centre for Forest Tree Technology of the Victorian Department of Natural Resources and Environment (NRE). NRE also supported an Executive Officer and provided secretariat services to administer the conference and facilitate the mid-conference field excursion.

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John Raison, Alan Brown, David Flinn
Canberra, October 2000

Introduction 1

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The concept of sustainable forest management (SFM) is not a new one. There is evidence that by the Middle Ages in Europe forests were managed by the rule that benefits were to be based on harvesting the interest, and not the capital, of the forest stock. Of course the early focus was on wood harvest and 'sustained yield'. In recent decades concern has broadened to cover the full range of goods and services provided by forests, and this change has been accompanied by increasing conflict over the relative priority to be given to different forest values where management has been for 'multiple uses'. Stakeholder interest and involvement has increased, and globally there is ongoing effort to develop a shared understanding of sustainable forest management and how it can be implemented in practice.

Commencing with the Earth Summit in 1992, there has been considerable global effort to better define sustainable forest management. This has focused the attention of all stakeholders and highlighted widely differing philosophies, understandings and expectations. The recently proposed concept of criteria and indicators (C&I) provides promise as an important communication tool, and as an underpinning for improved (adaptive) forest management. However, there are widely varying views on the role and value of C&I amongst stakeholders, and so far there has been little realization of potential benefits in terms of better forest management on the ground. There is a need to overcome a range of impediments, and to demonstrate benefits, in order to establish momentum in the application of C&I.

A key challenge lies in the adaptation of C&I that have generally been developed for national-level application (raising awareness, gaining political commitment, providing a basis for high-level reporting) to the forest management unit (FMU) level where most forest management decisions are made.

Addressing these issues is very timely, for as Bass (Chapter 3, this volume) notes: 'At a time when clashes between stakeholders are escalating, trust needs to be built up between the three main groups: government, market and civil society. A common and clear language is needed. C&I have real potential to provide this'. However, Bass also cautions against the growing mismatch between 'top-down' policies and local capacity to implement them, and supports a position of compromise and flexibility so that C&I can achieve local benefits for forestry practices and help secure outputs of forest goods and services.

A myriad of challenges face the successful implementation of C&I, and many are unlikely to be quickly resolved. These are of both a philosophical and a practical nature. Resolution of these impediments will require ongoing constructive dialogue and debate between stakeholders, and the critical evaluation of initial efforts to apply C&I at a range of scales in forests. These processes are at an early stage.

In August 1998 an international conference was held in Melbourne, Australia entitled *Indicators for Sustainable Forest Management: Fostering Stakeholder Input to Advance Development of Scientifically Based Indicators*. The meeting was organized by the International Union of Forestry Research Organizations (IUFRO), in collaboration with the Center for International Forestry Research (CIFOR) and the Food and Agriculture Organization of the United Nations (FAO), and hosted by the Centre for Forest Tree Technology of the Victorian Department of Natural Resources and Environment. The programme was developed by an international Scientific Committee.

The broad objective of the conference was to accelerate further development of scientifically based C&I for sustainable management at the FMU level. Additional objectives were (a) to encourage attendance and active participation of the full spectrum of stakeholders including policy makers, indigenous people groups, environmental non-governmental organizations, forest managers and labour organizations, and (b) to encourage discussions between those stakeholders and scientists. This was reflected in the conference motto 'fostering stakeholder input to advance development of scientifically based indicators'.

Specific conference goals included:

1. Review the state-of-knowledge for indicators covering the full range of criteria proposed under the various international processes concerned with sustainable management of natural forests and plantations.
2. Discuss and debate the adequacy of this science to meet stakeholders' expectations.

3. Obtain consensus on how scientific capability and stakeholder expectations can be brought together in pursuit of ongoing improvement in forest management.
4. Identify future research and development priorities for sustainability C&I.

After the conference, the invited keynote, overview and review papers for each of the seven generic sustainability criteria were peer-reviewed, revised and edited to form the basis for this book. The sustainability criteria covered are:

- Social and economic functions and conditions;
- Legal and institutional frameworks;
- Productive capacity;
- Ecosystem health and vitality;
- Soil and water protection;
- Global carbon cycles;
- Biological diversity.

We hope that the revised contributions assembled in this book will stimulate debate and provide guidance to the implementation of C&I to improve forest management.

2

Application of Criteria and Indicators to Support Sustainable Forest Management: Some Key Issues

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Criteria and indicators (C&I) have been proposed as new tools to help better define sustainable forest management (SFM) and to monitor temporal change in the condition and output of goods and services from forests. Whilst the broad rationale underpinning C&I seems straightforward, the application of C&I to improve SFM raises major challenges of both a philosophical (conceptual) and practical nature. A major issue is scale, and how to adapt C&I developed for national-level use to the finer scales of the forest management unit (FMU) where forest management decisions are made, and can be adapted over time if required.

Ongoing constructive dialogue between stakeholders is needed to progress the complex process of indicator development, testing and application. The goals and outcomes negotiated between those with legitimate interests in forests effectively become a working definition of SFM that takes into account local values and issues. Application of C&I clearly must be consistent with these. It is critical that stakeholder expectations are regularly tested against the scientific capacity to support cost-effective application of C&I.

Scientists have a key role in progressing implementation of C&I by synthesizing technical information and in integrating quantitative and qualitative information to make 'expert' judgements that have unknown uncertainty. This is not to say that scientists should not be vocal in stressing uncertainty and the need for further work to improve systems, but they have a responsibility to engage to assist initial application of C&I, recognizing that this will better define the actions required for evolutionary improvement.

We conclude that the application of C&I has considerable potential to improve forest policy and management, but that the science and application of sustainability indicators in forestry is very immature. Current capability and key issues still to be addressed are discussed by the authors of Chapters 3–20 of this volume, and cover the full range of sustainability criteria. Salient points and important outcomes from the Melbourne Conference, described in Chapter 1 of this volume, are also summarized here.

During indicator development and selection, the critically linked activities of monitoring and data evaluation must be addressed. If this is done in conjunction with stakeholder input to develop management goals and targets, C&I can be applied at the FMU level to underpin adaptive forest management.

1 Background to the development of C&I

There is increasing concern that forests worldwide are being degraded by the impacts of land use change, unsustainable harvesting, forest fires, climate change, disease, industrial pollution and other factors. The concerns were discussed at and reflected in outcomes from the 1992 United Nations Conference on Environment and Development (the Earth Summit). The set of non-binding but authoritative principles on forest management agreed at the Earth Summit included agreement by governments to develop sound criteria and guidelines for the management of all types of forests. This agreement was the seed for a number of global initiatives to develop scientifically based criteria and indicators (C&I) that could be used to support and monitor progress toward better (sustainable) forest management (Ramakrishna and Davidson, 1998).

The initial efforts were focused on national-level C&I, but there was a recognition that these needed to be adapted to finer scales that were more relevant to guiding and improving forest management actions on the ground. It has also been increasingly recognized that the effective application of C&I is of interest to a wide range of stakeholders, and that their input and support is essential to making new systems work. It is also clear that expectations, and understanding of the challenges associated with the implementation of C&I, differ markedly between stakeholders. Policy makers have shown enthusiasm for the concept, scientists have often urged caution, forest managers have concerns about complexity and cost, and conservationists have often been sceptical. There has also been concern that social and cultural issues are not being adequately addressed. Ongoing constructive dialogue between stakeholders will therefore be essential in order to progress the complex process of indicator development, testing and application. The Melbourne meeting that was a forerunner to this book provided an opportunity to share experiences and contribute to advancing the implementation of C&I in forests.

2 Sustainable forest management

SFM is an evolving concept that is of major interest to much of society. It is clear that there is no unique definition of SFM, but that it must reflect the goals and outcomes negotiated between those with legitimate interests in forests. In that sense, the goals and outcomes effectively become a working definition of SFM that takes into account local values and issues. There are likely to be few absolute requirements for SFM, with exceptions being avoidance of irreversible change such as species extinction. There will almost always be the need to balance social, economic and environmental values.

There are growing expectations that managers can demonstrate SFM by quantifying progress against goals and outcomes (targets). The application of C&I can help achieve this, provided that there is a shared view and agreement as to what indicators are used, the definition of targets, and how monitoring and review processes will occur. Stakeholders must be effectively engaged so that a genuine agreed position can be developed. If this is not done, ongoing conflict about forest use is likely to remain. Such engagement can result in development of a shared understanding of the benefits and risks of alternative forest management options. Importantly, this approach can provide a basis for ongoing review and improvement of management practices (Fig. 2.1).

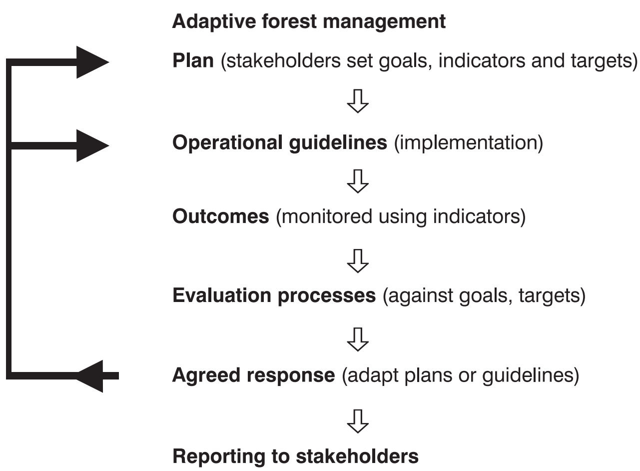

Fig. 2.1. The application of indicators, monitoring and evaluation procedures to underpin an adaptive forest management system.

3 Important drivers for the application of C&I

The application of C&I could have the following broad benefits:

- Raising awareness of and political commitment to SFM, with the result that governments will support the implementation of C&I, facilitate data collection, and take responsibility for international reporting.
- Provide a useful mechanism for reporting to both domestic and international stakeholders on the state and trend in condition of a nation's forests. Such information can be important to gaining community and political support for forestry activities, and especially access to native forests for wood production.
- Provide an important plank for the certification of forests as being sustainably managed, and thus for 'green' labelling of forest products and flow-on marketing benefits.

It is obvious that the latter two benefits require application of C&I at finer (management unit) scales. These benefits are what will encourage forest managers to commit to the application of C&I and to expend funds for monitoring, reporting, and research and development. Forest managers need to see clear benefits that justify investment in application of C&I. Excessive or sole focus on national reporting may create a lack of commitment from forest managers, with the result that the costs of data collection and reporting will fall largely to governments. The resultant financial burden will probably mean that little additional (indicator-specific) data will be collected and that reliance will be on analysis of remotely-sensed data that is often a poor surrogate for the criterion of interest (e.g. the use of forest area as a surrogate for biodiversity). Sharing of responsibility and cost for data collection between governments and forest managers is critical. Systems need to be developed to enable data to be aggregated and synthesized across varying spatial scales.

Equitable access to data, especially publicly funded information, is important to facilitating SFM. Lack of data sharing or transparency inhibits development of trust between stakeholders. Where information cannot be widely distributed, the reasons for this (e.g. cultural issues, commercial sensitivity) need to be clearly articulated and discussed.

C&I are only one of several tools that can help facilitate SFM. As described above, C&I need to be embedded in an Environmental Management System (EMS) for maximum effectiveness. The EMS supports the planning, implementation, monitoring and evaluation steps that are essential to adaptive forest management (Fig. 2.1). A critical component of the system is participatory planning and evaluation processes, that help to develop shared goals and agreed actions following the evaluation of collected data. Demonstration forests, that involve stakeholders in planning and review, and that examine forest management options, can also be a valuable way of engaging stakeholders and helping advance SFM.

4 Scale issues and the application of C&I at the forest management unit level

C&I can be applied at a range of spatial scales (e.g. ISCI, 1996). Early emphasis was on the development of national-level C&I (e.g. under the Helsinki and Montreal Processes) for the purpose of raising awareness, of gaining commitment, and to assist in measuring broad progress towards achieving SFM. Many of the national-level indicators are not sufficiently sensitive to be useful at the FMU level. For example, change in forest area is used as a Montreal Process indicator of biodiversity, but this indicator could not reflect important management-induced change in understorey structure and animal habitat (e.g. Noss, 1999). Similarly, soil and water values must be protected at the scale of the FMU.

Key requirements of any indicator are the ability to detect important change in forest condition, and the capacity for cost-effective application at an operational scale (Raison *et al.*, 1997). In order to help improve forest management, indicator selection must be linked to the essential steps of monitoring and data evaluation. These critical linked processes can be embedded within a forest management system, and provide the basis for adaptive forest management (Fig. 2.1).

In many cases, accurate measurements must be taken at a number of representative sampling points within the FMU. There is a range of strategies for achieving this in a cost-effective manner (Raison and Rab, Chapter 14, this volume). This fine-scale data can then be aggregated or extrapolated to larger (e.g. regional, state or national) scales.

To help address scale issues, Smith *et al.* (Chapter 11, this volume) suggest a linked and tiered approach to monitoring and inferring change in forest systems. This consists of monitoring at all sites for operational compliance (e.g. adherence to guidelines contained in Codes of Forest Practice), monitoring a limited number of sites using site-specific indicators to determine the effectiveness of recommended practices, and detailed monitoring and research on a few representative sites to provide the basis for validating or improving management guidelines.

5 Challenges in implementing C&I and a role for scientists

C&I are a new concept that has been developed to help provide greater clarity in defining SFM and tracking progress in achieving it over time. The criteria are used to describe the components of sustainability, and cover environmental (ecological), social and economic issues. Indicators measure various aspects of each criterion, and thus enable the effects of policy decisions and forest management practices on the state of forests to be monitored and reported. Whilst the broad rationale underlying the development and application of C&I seems relatively straightforward, many conceptual and operational issues still need resolving (Raison *et al.*, 1997; Smith and Raison, 1998). Of a technical nature, the major ones relate to:

- development of scientifically based indicators that meet stakeholder expectations and that can be applied at scales relevant to forest management decisions;
- defining cost-effective monitoring systems; and
- establishing evaluation procedures to enable interpretation and use of collected data.

The term ‘significant’ is often used in description of indicators, implying some critical or threshold value for change. The quantitative value for significance will depend very much on local context (nature of the forest ecosystem) as well as on management objectives (e.g. Burger and Kelting, 1998; Powers *et al.*, 1998). Thus there is a critical need to define threshold values for specific forest environments based on scientific understanding of processes and on input from stakeholders. This activity will require an integration of quantitative and qualitative information. There is a challenge for scientists, who prefer to work with quantitative data and to make largely objective judgements, to adapt their thinking and contribute to such integration. Some expert (but subjective) judgement is an essential part of this process, as is a willingness to draw conclusions that may have an uncertainty that is statistically undefined. This is not to say that scientists should not be vocal in stressing uncertainty and the need for further work to develop improved systems, but they need also to make use of what is known (synthesize the available scientific information) and to engage to assist initial implementation of C&I. Learning from doing is very important, with recognition that evolutionary improvement will be required. Ongoing research will improve the scientific base of C&I, and field testing will help identify where the science base is either too weak or the indicator is impractical.

The potential benefits to forest management and thus to broader society of successful implementation of C&I are large, and worthy of major scientific investment.

6 Key issues and actions needed to enhance indicator development for each criterion

6.1 Socio-economic values

There is an increasing and strong recognition that social impact assessment needs to be more widely applied in forestry as part of planning for SFM. Social values will change over time, and indicators must be capable of accommodating these changes. Whilst science can inform and guide indicator selection, it does not decide on what is best; stakeholders must collectively define what is to be sustained and how that should be assessed. Easy access to common information is essential to participatory planning, and this facility needs to be improved in many jurisdictions.

Social and economic indicators need to make sense at small scales, and to relate to local communities. Improved methods are required for exploring and agreeing on 'trade-offs' between social, economic and biological values. These methods are critical to facilitating participatory planning.

The existence of legal and institutional frameworks in which there is sufficient commitment and capacity is critical to supporting SFM over time. In many instances, legislation still lags behind the rapidly escalating commitments needed to achieve SFM. Enforcement can be important to ensuring actions needed to achieve SFM (Eeronheimo, Chapter 8, this volume) but so also are incentives.

To date many indicators in this area are based on management inputs, but as McCool and Stankey (Chapter 6, this volume) stress, the focus needs to move towards outputs which reflect the degree of progress towards achieving objectives. McCool and Stankey also stress the need for reasonably comparable indicators, so that comparisons can be made over time and to some extent between areas (although there can be good reasons for using different indicators and threshold values in different forest settings). Qualitative indicators may be especially useful as socio-economic measures.

There is clearly considerable benefit from ongoing sharing of experience and information between countries engaged in C&I initiatives. Substantial steps have already been taken in this direction, for example in Europe. This will assist with capacity building, clarification of terminology and broad harmonization of approaches.

6.2 Productive capacity

The definition of productive capacity almost always causes some confusion. In terms of the C&I framework, it refers to the actual productive state of the forest which reflects both inherent site potential (especially soil and climate) and the way the forest has been managed. Forest productivity cannot easily be used as a measure of soil fertility because of the significant confounding caused by management factors (e.g. weed management, site occupancy, pest control, genetic composition). A separate sustainability criterion deals with the protection and maintenance of soil values (fertility).

Increasingly there is emphasis on the wide range of goods and services that forests provide in addition to traditional timber values. Methods for measuring timber values and for predicting future productive potential are much better developed than those for measuring non-wood values. Indicators that relate to area of forest available for wood production, adequacy of regeneration and the sustainability of wood harvest, are adequate to cover timber production aspects of this criterion. A separate set of indicators is needed for non-wood values, and these indicators probably need to be flexible and adaptable into the future.

The relationship between rates of production of wood and non-wood goods and services is generally not well understood and is a strong limiting factor in forest planning. There are strong links between productive capacity and other sustainability criteria. The conversion of natural forests to simplified (usually monospecific) plantations may adversely impact on the production of non-wood products in some regions. The consequences of this need to be evaluated in light of the benefits of plantation creation.

Information on productive capacity is generally best for plantations and for even-aged forests. There is often a need for better information on tropical native forests, uneven-aged mixed species forests and for minor species in temperate regions. The difficulty of measurement and forecasting growth is much greater in the latter group of forests.

Regeneration is a key process for the maintenance of productive capacity in native forests, but indicators for this are either missing or inadequate in the various C&I schemes. This issue needs to be addressed, and must include a basis for setting minimum standards for regeneration success.

Remote sensing provides an increasing potential for monitoring forest condition (structure, biomass, species composition, regeneration, health, etc.) but further research and development involving ground-truthing is required to confirm utility in specific forest types.

6.3 Health and vitality

Forest health clearly must be defined in terms of the management objectives for a particular forest (Innes and Karnosky, Chapter 13, this volume). However, forest health is a difficult concept and this makes the task of developing C&I in this area very challenging. Measures should either reflect an important ecosystem process (e.g. soil acidification, canopy photosynthetic rate) or forest condition (e.g. leaf area, decay of bolewood). This criterion and associated indicators relate closely to the criteria covering biodiversity, soil and water, and productive capacity. There are two types of indicators – those measuring stress (e.g. critical loads of pollutants) and those measuring health (condition). Indicators reflecting change in forest condition are more desirable, especially those that reflect change that is either irreversible or very difficult to reverse.

Innes and Karnosky (Chapter 13, this volume) propose a range of indicators that reflect levels of environmental stress (critical loads of pollutants, soil N saturation or negative balance of important elements, presence of exotic pests and pathogens) or impacts of stress (wood quality problems, proportion of salvage logging, dead trees, reduced genetic diversity). They suggest that with further work these could become useful monitoring tools. The proposed indicators are valuable in that they could be applied at a range of spatial scales. Innes and Karnosky make the point that it is impractical to suggest a set of indicators that apply to all forests that have diverse management objectives, but they suggest that indicators can be derived for particular sets of activities such as nature conservation, timber production or recreation.

One important approach to monitoring forest health is the use of remote sensing of canopy condition (Datt, 1999; Stone, 1999; Stone *et al.*, 1999). This relies on measuring canopy area and the chemistry and physiological activity of foliage, as indices of ecosystem processes affecting forest functioning. Providing satisfactory calibration of remotely sensed data with ground measures of forest health can be obtained, new remote sensing approaches will allow monitoring over extensive areas at modest cost. It is also important to develop some indicators that provide early warning signs of change in important ecological processes (e.g. leaching of soil cations in polluted environments) before there is any observable or easily measurable effect on crown conditions.

Beese and Ludwig (Chapter 12, this volume) propose a set of indicator types which when linked can provide useful guidance for SFM. The indicator types are:

- Analytical – describe change in the state or functioning of components of an ecosystem (e.g. N concentrations in foliage). They give only broad guidance as to the reasons for change or its likely importance.
- Compound – combine a range of measures or observations to give greater interpretative power, e.g. biological availability of a limiting nutrient, or of a toxic substance, in the soil.
- System – which cannot be measured or observed directly, but are derived from other system properties (via the combining of analytical and compound indicators). Examples of measures derived are stability, resilience and the potential for forest development.
- Normative – provide evaluation of high-level components such as ethic, social, economic or political factors. They provide information about the quality of a system and its development for human needs. By setting a threshold value for quantitative change, an analytical indicator can be transformed into a normative one. This approach allows indicators to be developed to support multifunctional forest use.

6.4 Soil and water values

Maintaining soil and water values is critical to the regulation of most ecological processes in forests. Despite this importance, it is a major challenge to identify generic and relatively simple indicators that reflect key soil and water values given the complexity of processes involved, and the fact that natural spatial and temporal variability is usually high. It is not surprising, therefore, that limited progress has been made in establishing robust soil and water indicators, and in generating spatial data sets that can form the basis for assessing sustainability.

The need to assess soil and water change at fine scales presents a further challenge. Such assessment is necessary because local effects have major impact on many important ecosystem processes (e.g. soil compaction on a logged area affects seedling establishment and growth, the generation of overland flow and erosion potential, as well as soil biodiversity and carbon dynamics). There is interest in water values at a range of scales – local deterioration of water quality adversely impacts on stream biodiversity and local communities, whilst catchment-scale changes in water yield and quality are critical for urban communities.

Raison and Rab (Chapter 14, this volume) proposed the following broad set of soil indicators for application at the FMU scale: organic matter, acidity and base status, density and erodibility. They outlined a set of field measures related to these, but emphasized that 'local' evaluation of indicators was a critical next step.

Roberts (Chapter 15, this volume) has proposed a set of indicators for water flow and for water quality that can be applied at catchment or finer scales. For the quantity and timing of streamflow these relate to: forest cover and its change; fraction of forest cover which is conifers, broadleaf evergreen or deciduous forest; forest growth rate; and adjacency of forest to streams. For water quality the suggested indicators were: fraction of area occupied by roads, skid trails and log landings; fraction of stream length protected by adequate riparian buffers; density of stream crossings and contiguities (streams and roads adjoining without adequate buffer strips); and occurrence of macro-invertebrates. These indicators are based on a large body of scientific data collected in a wide range of forest ecosystems. Despite this, they require local evaluation and/or calibration before they can be confidently applied.

A stratification of the forest estate, based on the potential risk to soil and water values imposed by management practices, can be used to devise a strategy for monitoring change in these values on selected management units. Further stratification within the managed (e.g. harvested) area can also improve the efficiency of sampling and assist interpretation of the importance of measured soil change (Raison and Rab, Chapter 14, this volume). A temporal framework for monitoring should take account of contrasting rates of change in soil or water properties after management interventions. A strategic approach to monitoring that focuses on representative high-risk situations, and that tests the effectiveness of practices used to mitigate any potentially adverse effects of management, is argued to be cost-effective (Raison and Rab, Chapter 14, this volume).

Monitoring of change in soil properties is rare in operational forestry, and the situation is only slightly better developed for water values. For most jurisdictions, operational application of soil and water indicators is not yet possible. The goal must be to move from the common use of 'input'-based measures (e.g. compliance with planning guidelines) to outcome-oriented indicators that can provide a basis for adaptive forest management that meets soil and water objectives. As a move in this direction, semi-operational testing (case studies) in important representative forest types would be an effective way of evaluating interim indicators and developing approaches for their operational application. Again, learning from practical experience will greatly facilitate the process.

6.5 Carbon balance

Forests and forest management practices significantly affect global carbon (C) cycles. The importance of these continues to escalate as the international community wrestles with a response to greenhouse concerns. The estimated global carbon stock in forests is roughly equivalent to that in the atmosphere. Significant annual net carbon flux globally is associated with deforestation in the tropics, increasing forest cover at higher latitudes, and harvest for fuelwood and forest products. At 'local' scales many forest management decisions affect C storage in the forest (e.g. preserving forests with high biomass content, afforestation, lengthening of rotations, retention of slash, control of fire). However, there is a finite capacity for C storage on any hectare of forest, and the greatest potential for forests to make an ongoing contribution to net greenhouse gas emissions is via the substitution of wood for fossil fuels (forest bioenergy) or by substituting wood for other materials that have higher carbon dioxide emissions associated with their manufacture.

There are no simple methods for accurately estimating net changes in forest ecosystem carbon storage at either the stand level or at larger scales. Small temporal changes in carbon stocks are difficult to detect against a large background pool in many forests, but even a small change occurring over large areas results in very important change in terms of global carbon cycles. Kirschbaum (Chapter 16, this volume) concluded that assessing changes in forest carbon stocks at a regional scale requires a combination of methods: ground-based measures, including inventory and harvest removals, growth modelling, remote sensing, and atmospheric trace gas studies. It is likely that the intensive efforts being put into carbon accounting internationally, in response to needs of the Kyoto Protocol, will generate some simple rules (indicators) that can be used to track change in forest carbon stocks at a range of spatial scales. Such rules are particularly needed to describe temporal change in below-ground carbon pools in roots and soil organic matter.

In common with other criteria, there is scope for substantial international collaboration to refine indicators that reflect the contribution of forests to global carbon cycles. The current activities linked to the Kyoto Protocol should be exploited for this purpose. There are clear linkages (and benefits of collaboration) between efforts on carbon budgeting and activities under the criteria covering productive capacity, and soil values (organic matter and its links to soil fertility).

6.6 Biodiversity

Biodiversity is highly scale-dependent, and differing kinds of indicators are needed for ecosystem, species and within-species (genetic) components. The long-term impacts of forest harvesting on biodiversity are poorly known in most parts of the world, and this is a major problem limiting indicator development and application. Given these uncertainties, a strong adaptive management approach is advocated that involves partnerships with stakeholders in the planning and evaluation stages, and that has a clear link to the legal framework. Mechanisms are needed to share data, except where legal considerations restrict this, in which case this needs to be explained at the beginning of the process. Critical issues are synthesizing available information, clarifying scale issues and determining thresholds.

Finegan *et al.* (Chapter 17, this volume) conclude that a practical approach to developing indicators of ecosystem-level diversity is to focus on forest 'types' defined according to an understanding of ecological differences between them. This approach assumes that conservation of representative samples of forest types will conserve not only those ecosystems, but also (through the 'coarse filter' principle) most of the species which make them up. In some parts of the world, there is still a basic need to develop vegetation classification systems at appropriate scales as a first step to better forest planning and biodiversity conservation. Finegan *et al.* also specify that the proportion (and degree) of modified forest types needs to be known – this raises the difficulty of defining finer-scale (more precise) measures of biodiversity that relate to changing levels of disturbance.

Kanowski *et al.* (Chapter 18, this volume) make the important point that achievement of biodiversity conservation objectives relies on the success of management both within and outside reserves. They stress that the success of reserve or off-reserve management cannot be assessed until biodiversity conservation objectives have been agreed and articulated on a bio-regional basis. Clearly, effective processes for stakeholder involvement are required to achieve this. The relative importance to biodiversity conservation of reservation compared with off-reserve management varies dramatically around the world, with reservation more significant in areas where a greater proportion of forest land is in public tenures. However, forests managed for multiple uses are almost always important for biodiversity conservation. Indicators of the success of management in achieving biodiversity conservation objectives will be different in reserves and off-reserve forest because of the different contributions that each make.

Loyn and McAlpine (Chapter 19, this volume) stress that forest fragmentation and its biological effects at the landscape scale are very complex issues that are not very well understood. For example, thresholds in metric values (e.g. patch size) for specific biota are generally unknown. As an interim approach they suggest that management aims to create a variety of structures and spatial patterns to help cope with the many uncertainties of biodiversity conservation. They emphasize that at this time indicators cannot be perfect or restricted to a definitive set, but they need to reflect a shared and ecologically sound understanding of the condition of the forest. Two types of indicators are proposed: (i) landscape metrics (assessed remotely) that are spatial measures of whether aspects of landscape pattern critical for particular species assemblages have been maintained; and (ii) species-based indicators that encompass groups of species with varying life histories and scales of movement. The authors propose a flexible approach to indicator use, consistent with management objectives in particular forest types and locations.

7 Conclusion

Use of C&I has considerable potential to improve forest policy and management, but the science and practical experience of applying sustainability indicators in forestry is very immature. During indicator development and selection, the critically linked activities of monitoring and data evaluation must also be addressed. If this is done in conjunction with stakeholder input to develop management goals and targets, C&I can be applied at the FMU level to underpin adaptive forest management, as summarized in Fig. 2.1.

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3

Policy Inflation, Capacity Constraints: Can Criteria and Indicators Bridge the Gap?

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The 1990s may go down in the annals of forestry as the decade when foresters – and just about everybody else – attempted to define, or to prescribe, the elements of sustainable forest management (SFM). Such efforts have painted rich canvases of multiple forest values and actors, particularly through sets of criteria and indicators (C&I). This chapter discusses the context for the further development of C&I and proposes how we can maximize their utility.

The chapter proposes a shift in emphasis from setting goals in terms of forest area or forestry practice, towards goal-setting in terms of the *security of specific forest goods and services*. Special attention must be given to the security of multiple goods and services at the household level, especially for the world's one billion poor who are dependent upon forests to a significant degree. C&I must accommodate their values. Also, foresters will have to think more broadly than forests alone for achieving security of forest goods and services: imports and substitutes, and trees on farms, are often as important as domestic production in plantations, multiple use forests and set-aside natural forests. The challenge for choosing between these calls for C&I that can handle the whole land use spectrum, as well as assist comparison with substitutes.

In facing the future for forests and stakeholders, it is suggested that forest management systems should be based on precaution, learning, adaptability and resilience. It is further suggested that C&I are integral to such systems, helping us to make the transition between where we are now, and where we want to go.

However, in the development of C&I, reflection is needed on the creeping trend towards 'policy inflation'. A forest manager can scarcely begin work until he or she has conducted impact surveys and consulted with many groups. Forest managers almost invariably find that policies are over-designed: they do not quite fit with local circumstances, and implementing them does not build management capacity to achieve local goals. Perfectionist approaches to C&I can be the enemy of the good. Instead, C&I must facilitate compromise and adaptation.

Recent sets of forestry C&I tend to have been 'quick fixes' and centralized solutions, and thereby may be predisposed to top-down control and implementation, whether intended or not. They might be contrasted with, e.g. C&I for organic agriculture, which have been slower to evolve and based on mutual recognition of many local standards.

The chapter concludes by urging the application of forestry C&I in the other negotiation processes that might be expected to have significant impacts on the security of forest goods and services, and notably:

- *ecolabelling initiatives* need to urge parallel consideration of the environmental and social standards which should be expected of substitutes for forest goods (lest forestry should suffer from its noble task of defining SFM C&I);
- the *environmental conventions* are fast becoming means to support or constrain funding to forests, and must surely become better informed about how good forestry can produce local and national goods and services (and not just global services). This is especially the case for carbon offsets; and
- the *finance and insurance sectors*, which need to open the doors to investment in SFM and close the doors to asset-stripping approaches.

1 The future for forests

The last decade has resounded with heated debates about 'saving the forests'. Targets have sprung up for protecting forests and for afforestation, most of them expressed in terms of forest area. Some area targets are now qualified by criteria and indicators (C&I) for the quality of forests or their management. Such targets often reflect little more than planners' dreams.

For what purposes are the forests being 'saved' – and for whom? Surely what counts is not the *area* of forests but the *forest goods and services* which are produced and sustained, and who has access to them? Debates on forests might progress better if we are clearer about what goods and services are required, now and in the future. Table 3.1 suggests the broad categories.

Goal-setting for forestry should be more specific about the need for a minimum *security* of specific forest goods and services. The concept of food

Table 3.1. Key forest goods and services.

Goods and services from forests	Local/household benefits	National benefits	Global benefits
1. Wood products	✓		
2. Non-wood products	✓		
3. Watershed functions	✓	✓	
4. Soil protection/nutrient cycling	✓	✓	
5. Wind and noise control	✓	✓	
6. Microclimate moderation	✓	✓	
7. Recreation and tourism	✓	✓	
8. Cultural and spiritual values	✓	✓	
9. Sense of place	✓	✓	
10. Landscape and aesthetics	✓	✓	✓
11. Biological diversity/potentials	✓	✓	✓
12. Climatic stability		✓	✓

Developed from Segura *et al.* (1997).

security certainly provided a way forward in the 1960s and 1970s for the agriculture sector. It helped it to ensure that food needs at household, national and regional levels were recognized, integrated and achieved.

Assuming that such goals of forest security can be agreed (and I return to this assumption later), a key question is *what production and market systems can best meet them?* The answer is not solely to be found in forestry textbooks: increasingly, other land uses will also be well-placed to produce the required goods and services:

1. There is a wide *spectrum of land uses* from which forest-related goods and services can be derived, and not just natural forests and plantations (Fig. 3.1). In many, food production is the key and food security goals will need to be integrated. For example, most timber is produced on farms in Pakistan, although this is barely recognized by the authorities (Ahmed and Mahmood, 1998). The security of biodiversity assets is associated with very specific parts of the spectrum.

2. At the national level, choices can be made to *import* forest goods and services, rather than to produce them domestically (everything from importing timber to paying other nations to conserve carbon stocks in forests). The key issue here is the ecological and social 'footprint': whether the importing nation is sending out signals that encourage good forest management abroad – or alternatively is triggering asset-stripping.

3. Choices can also be made to *substitute* for forest goods and services, e.g. by employing metals or plastics. Price and product specification have dominated such decisions so far. We are lacking ways to compare the social and environmental impact of different production processes for unlike products. If forestry becomes transparent about its production methods, but other industries do

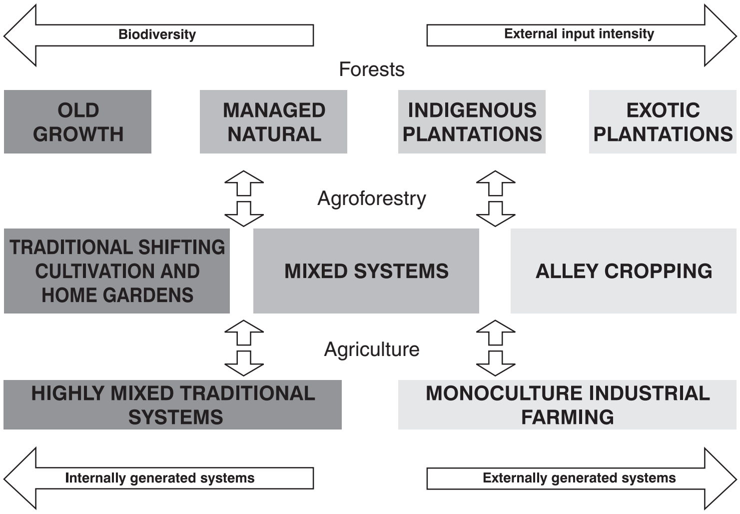

Fig. 3.1. Land use spectrum.

not, we may witness an incompletely informed public turning away from some forest products.

4. Finally, changes in technology can redefine the possibilities for all the above. Sayer and Byron (1996) show how technological developments in plywood, veneer and MDF (medium-density fibreboard) manufacture have already had enormous influences on the forests of South-East Asia. The ways in which forests are managed to produce these products favours uniformity, and not the multiple uses being sought today. This trend is set to continue: new technologies such as Plato and Valwood, which may be able to provide hardwood characteristics from softwood, could accelerate the trend to plantations.

The net result will be a new spectrum of land use types. The main 'colours' in this spectrum will be: plantations/intensively managed forests; natural forests of primary characteristics, set aside for non-consumptive purposes; natural forests managed for multiple consumptive purposes; and small farm landscapes with trees (Bass, 1997).

1. *Plantations or very intensively managed forests* – focused on wood production. There is a major trend for plantations to become the predominant production system for fibre. Global plantation area doubled from 85 million to 180 million ha between 1980 and 1995; and is expected to double again between 1995 and 2010 (Food and Agricultural Organization of the United Nations (FAO), 1997). This trend comes about because:

- remaining 'primary' natural forests are comparatively inaccessible to economic utilization. Unit costs of natural forest wood will go up, especially with demands for environmentally sensitive production;

- competition for land from agriculture and tree crops will render wood production from natural forests comparatively unproductive in societal terms. FAO (1995) predicts a significant amount of forest conversion for farming, in spite of agricultural intensification;
- market and regulatory pressure will result in natural forests focusing on non-wood values, such as biodiversity and watershed conservation;
- technological dependence on certain natural forest woods for specific purposes will decline;
- investment in genetic and silvicultural work is greatly increasing the productivity of plantations. Unit costs will go down; and
- globalization and economic liberalization will lubricate the above processes, by allowing the movement of capital and technology into new plantation enterprises.

The challenge will be to ensure that plantations are integrated into a landscape which can provide other goods and services (for which local government and community rules may play key roles). Or, if plantations are very large and dominate the landscape, developing multiple-use management systems to ensure plantations themselves provide other benefits (for which the right mix of voluntary efforts and external controls may be needed). Both organic agriculture and forest certification systems now seem to be realizing that certain trade-offs, and the integration of goods and services, can be made only at the landscape level, rather than within the individual farm or forest management unit (FMU).

2. *Natural forests of primary characteristics.* These will increasingly be protected for non-consumptive use of environmental services, notably carbon storage, biodiversity, wilderness and watersheds. The challenge will be to identify the minimum set-aside required to sustain each of these services, and to agree protocols to pay for them. A continued emphasis on global legislation and/or markets can be expected. Despite a spate of protected area establishment in the 1960s to 1980s, there has been a slow-down, due largely to political and institutional constraints. There are international guidelines for biodiversity protection, e.g. 10% of land area, but these are only negotiating points. Sayer *et al.* (1997) describe how new knowledge (from both North America and the tropics) suggests that the areas required to protect representative ecosystems are far smaller than was once thought.

3. *Natural forests managed for multiple purposes*, including consumptive uses. Many of these should remain under the control of local forest-dependent people, to meet livelihood needs first. This will lead to highly *mixed landscapes*. As Filer and Sekhran (1998) point out for Papua New Guinea, the people whom outsiders often like to define as ‘indigenous forest people’ actually think of themselves as forest gardeners and developers. For commercial forestry, the challenge will be to develop management systems that optimize the production mix, e.g. that understand the trade-offs between different wood production levels, livelihood needs and biodiversity; and to create the incentives for these kinds of management. We need to be more pragmatic about the possibilities for conserving some elements of biodiversity within forests managed for timber. Jerry Vanclay of the Center for International Forestry Research (CIFOR) has observed how the British Navy's strategy for protecting small islands (which we may consider analogous to nature reserves) was not by manning large fortresses on them, but by ruling the seas (analogous to managing forests between the reserves). However, we cannot assume that the biodiversity elements which people value most can always be sustained in managed forests. An adaptive, highly monitored approach is needed in natural forest management. Indeed, in circumstances of incomplete information, every managed forest needs to establish a hypothesis about product mixes, and include ways to monitor change, notably the approach of possible thresholds beyond which biodiversity may not recover.

4. *Small farm landscapes with trees*. Many local people who are short of forest goods and services *create tree resources* within predominantly urban or agricultural landscapes (Arnold, 1997). An example from the Niger is shown in Fig. 3.2. Although so far this has proven difficult to monitor by foresters, these tree resources are significant for security of forest goods and services. For example, while many outsiders have been worrying about deforestation rates in Kenya, net tree cover appears to have increased due to farm planting. The challenge will be to ensure people have the rights, resources and incentives to do so, and to avoid much of the 'top-down' imposition of agroforestry models which we have seen until recently.

The notion of SFM has to apply to all colours in this land use spectrum. No one form of land use is intrinsically more sustainable than the others. They play different roles in the search for security of forest goods and services.

In short, the arena for change encompasses far more than the current forest estate, as non-forest land will come under trees, and as substitutes for forest benefits are developed. Questions of efficiency, comparative advantage, equity and sustainability arise when making choices. Commonly accepted language – and new means of assessing the options – will be needed. This may be where the current preoccupation with C&I ultimately proves its worth.

2 The future for forest interest groups

The language used by forest interest groups to articulate forest issues can be startlingly diverse – reflecting the huge diversity of groups themselves, their understanding and experience of forests, and the range of values which they ascribe to forests.

What defines the relationship between an interest group and the forest? Firstly, there are the different *benefits* that people seek from forests (Table 3.2).

Fig. 3.2. These plantings of *Acacia holosericea* near Maradi, Niger are providing multiple benefits for local communities. The trees are stabilizing the soil against erosion in situations where goats have removed much of the native vegetation. The acacias also produce seed that can be stored and eaten when drought reduces other food supplies. Environmental and social indicators are needed to quantify the efforts of such rural tree plantings.

Then there are the different *means* by which the group invests in and manages forests to realize these benefits. Recent work in Africa shows that local people will manage a forest if they:

- believe the forest is theirs;
- have both the desire and the incentives to manage it; and
- have adequate rights, access to resources, rules and organizations to do so (Gill Shepherd, London, personal communication, 1997).

Similar conditions can be observed for interest groups other than local people. The difference is often that some groups (notably corporations) have the necessary rights – or are effective at making claims to them – and have access to resources, but local people often do not.

The weakness of forest-related institutions, and differential access to decision-making processes, mean that different interest groups' values are often not reconciled. Instead, certain groups' values are becoming more predominant: groups who are often furthest away from the forests themselves, yet who are privileged to be closest to centres of policy and market power, such as corporations, retailers, environmental groups and government bodies

Table 3.2. Forest values depend upon who you are.

Forest interest groups	Approx no. (million)	Important forest values
Urban people	2500	Wood, non-timber forest products (ntfps) Water, climate moderation
Rural poor/landless	1000	Food/fibre/health components of livelihood Support to farm systems
Shifting cultivators	250	Much of livelihood Spiritual and cultural values
Forest communities	60	Sole means of livelihood Spiritual and cultural values
Agribusiness	10	Land, water Support to farm systems
Oil/mining business	7	Minerals
Logging business	5	Timber
Retailers of forest products	3	Timber, ntfps, environment
Ecotourism business	3	Landscape, recreation Biodiversity, culture
Environmental groups, scientists	1	Carbon storage, climate Biodiversity, culture

Developed from WCFSD (1999).

(Ahmed and Mahmood, 1998). For example, environmental groups and some large retailers (notably the organized buyers' groups who intend to deal only in certified timber) appear dominant in defining SFM. There are some dangers that local social values may be submerged in this, and value accrue disproportionately to groups who are furthest from the forest.

Special attention should therefore be paid to local forest-level stakeholders, especially if poverty and forest-related clashes are to be alleviated. As the World Commission on Forests and Sustainable Development (WCFSD) contends, 'over 1 billion people (about 20% of the world's population) depend either wholly or to a significant extent on forests, woodland or farm trees for their subsistence needs and/or livelihood' (WCFSD, 1999). In the 32 countries which suffer food insecurity, poor people depend upon forests for many regularly utilized foods, for crisis or famine foods, for firewood to cook with, for building, for medicines and nutrients to increase the nutritional impact of foods, for grazing, for genetic resources to improve food crops, and for protection of farms from wind or for shade (Mayers, 1997).

Global population will double by 2050, mostly in developing countries and amongst poorer groups. If current trends continue, many people may not have adequate access either to natural forests, or to resources to plant their own trees, or to the market to buy forest goods and services. As a representative of an indigenous group impressed upon a hearing of the WCFSD:

> Dayak people have lost access to lands that have been part of our livelihoods for thousands of years. We have lost access to woods, to rattans, and to many other forest products. It is not fair if the government decides that only timber companies can cut trees and trade the wood.
>
> (WCFSD, 1999)

These kinds of clashes are escalating. In a recent global survey of forest companies, 80% declared they had had conflicts with local or national groups (International Institute for Environment and Development (IIED), 1996).

There are key challenges associated with the ‘opening up’ of forest policy and decision-making processes to the various interest groups:

1. *Relationships and trust need (re)building between interest groups.* Things might improve if the actors on the forest scene reform their relations in ways which could be described as a ‘triangle’ of government, market and civil society bodies (Fig. 3.3), instead of ‘concentric circles’ around government. Such reformation is, in fact, taking place. National forest authorities are fast becoming intermediaries between global initiatives and parochial interests. Many roles of the forest authority are being passed to the private sector, which not infrequently has more resources and longer planning horizons than government. Civil society groups, from local chipko-type movements to well-connected international non-governmental organizations (NGOs), are challenging the supremacy of forest authorities. As a result, glaring lights have been shed on formal forest law, which may now be exposed as little more than historical anomaly and in dire need of change. Forest authorities admit (if tacitly) that formal policy is now very often less significant

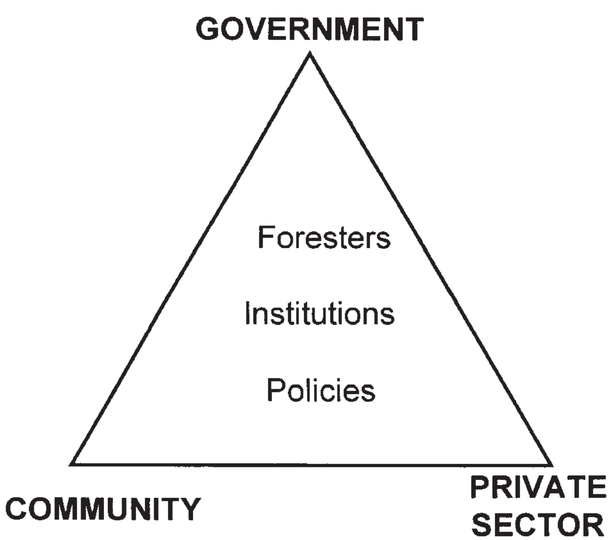

Fig. 3.3. Relationships between the main forest interest groups.

than other sectors' policies, macro-economic policies, or civil society/market 'soft' policies.

2. Trust requires the development or acceptance of common *language*. The 1990s may go down in the annals of forestry as the decade when foresters – and just about everybody else – attempted to define, or to prescribe, the elements of SFM (Bass, 1997). Such efforts have painted rich canvases of multiple forest values and actors, particularly through sets of *principles, criteria, indicators and standards*. Some are the product of just one group (e.g. industry codes of practice), while others have been negotiated amongst several groups (some (inter)governmental and civil society initiatives). Much of this effort could be explained away as attempts to control language to suit each group; however, as the initiatives have evolved, many interest groups have come to respect others' views. In many ways, they have served as massive mutual training courses.

3. Trust also requires the development of *communications and participation*. There is more or less an inverse relationship between public trust in an institution, and demands for participation. There has been quite a lot of confusion about where and how participation should be conducted. Exhortations, e.g. through Agenda 21, to conduct all affairs 'with the maximum possible participation' are difficult to respond to, although they reflect today's general lack of trust in forest institutions. The notion of a 'national forest forum' is now commonly advanced, but there is little experience of how to run them. The Canadian Round Tables in forestry, environment, and pulp and paper, have been assessed, and found to be significant in improving consultation across institutional barriers and generating ideas. However they are still generally marginal to core policy making, as they have not shifted those barriers (Bass *et al*., 1995). The participation in C&I processes also appears promising in that it has generated ideas and partnerships beyond the task at hand. Certification itself is one means for building a new basis of trust, albeit one which depends upon formal verification and accountability.

4. *The nature of the power structures* which lie behind many forest problems, and the inequalities which separate groups, must be addressed more overtly. All interest groups' needs and perspectives should at least be properly considered in forest-related decisions. We hesitate to use the term 'stakeholder', as it implies that a person or group has the power to make real inputs into decisions – which masks the fact that some groups effectively have no voice, or means to pursue their undoubted interests. This problem must be addressed if 'stakeholding' approaches to decision making are to work. An understanding of 'who counts most' is crucial to improving interest groups' relations, in terms of who has most dependence on forests, understanding and knowledge of forests, existing rights, and cultural links to forests. Here, the work of Colfer and others at CIFOR is central.

5. Reflection is needed on the creeping trend towards 'policy inflation'. There is a growing mismatch between policies and local capacities to implement them. We have noted that forestry is no longer merely circumscribed by national forest policy and legislation. It is now faced with global and local regulation in all its aspects – economic, social and environmental – and changing market requirements in terms of procurement, quality standards and demands for accountability. A forest manager can scarcely begin work until he or she has conducted impact surveys and consulted with many groups. Forest managers almost invariably find that policies are over-designed and do not quite fit with local circumstances, and if taken literally will lead to an approach akin to circus dogs jumping through hoops – rather than really building management capacity to achieve local goals. Perfectionist approaches can be the enemy of the good. They may especially prejudice against small local groups who cannot (and probably should not) meet high standards of commercial forest practice and transparency.

3 Facing the future: forest management systems that build in precaution, learning, adaptability and resilience

Where once forestry was about matching a relatively predictable set of demands to given site/climate conditions, the challenge is now to make decisions that meet multiple needs in an environment of political, market, social, technological and climatic change and uncertainty – and, moreover, in circumstances of incomplete information.

SFM will be increasingly *information-intensive*, but that does not mean it should be information-profligate. Information now firmly joins land, labour and capital as the classic production factors. Just like the other factors, information should be applied to the right degree: enough to get the product right, but not so much that it is too costly.

SFM will also be more *future-orientated*: forest debates and decisions need to be informed far more by scenario development than they have been. A precautionary approach is also indicated.

Complex social, economic and ecological systems which have been able to face the future tend to have certain basic *characteristics of resilience and adaptability* (Box 3.1). They are built upon highly connected mechanisms for communication and feedback. (However, they are not deterministic; in other words, as Ruitenbeek and Carter (1998) point out, such systems will still generate 'surprises'.)

Many of the elements of resilient, adaptive systems can be built up through traditional or proven local processes, and research should be seeking these out: e.g. 'traditional' resilient technology that has survived change, community fora and rules for dealing with change, observation of traditional indicators of change, conserving genetic resources, and other means to reduce vulnerability.

Box 3.1. Resilient, adaptive systems (Source: Bass (1996)).

Some common features of resilient and adaptive systems, taken from studies of social and ecological systems, may suggest some ingredients for future forestry systems:

1. High connectivity and extensive information flows within and outside the system:

- timely information on the current internal and external situation and (imminent) changes;
- abilities to make links, alliances and partnerships; and
- links and information that lead to rearrangement of the system's own highly connected components/hierarchies.

2. High degrees of interaction and participation in:

- decision/rule making;
- implementing agreed actions;
- cost–benefit sharing;
- monitoring and reviewing results; and
- innovation and technology development.

3. Emphasis on learning within the system, and incentives for this:

- continuous-improvement, double-feedback management approaches, i.e. learning about learning;
- future orientation – anticipating future influences and options;
- adopting the precautionary principle;
- ability to monitor and analyse change;
- distinguishing trends/events to which the system should adapt, from those to be buffered against, from those which are irrelevant; and
- ability and incentives to innovate.

4. Emphasis on local capacity and control:

- multiple skills;
- flexible teams and partnerships;
- diversity of perspectives encouraged and used;
- subsidiarity and dispersed control (more robust than centralized); and
- triad of government, market and civil society.

5. Information-intensive, diverse production methods:

- diversity of (local) inputs producing multiple goods and services.

These characteristics help to confer the ability to deal with change in the most appropriate way, i.e. to:

- buffer against immediately harmful change, or
- influence the outside change environment itself, or
- adapt where necessary.

This can be complemented by new information and communications technology which allows networks to form and share timely information on change and responses. The key to all is to install or strengthen learning systems. The Economist Intelligence Unit's Vision 2010 survey of 350 senior executives concluded that, with so much uncertainty and change in the world, organizational attributes considered useful in some years may not be fruitful in others; instead, the *rate at which organizations learn* may become the only sustainable source of comparative advantage.

4 Criteria and indicators: a bridge to the future?

To summarize the above discussion, I have suggested the need for forestry to develop so that it:

- is focused on goals to secure specific goods and services;
- accommodates any land use which can produce these goods and services;
- puts livelihood needs first;
- encourages markets for certain services produced (anyway) by good forestry, thereby improving commercial viability;
- allows informed comparison of forest goods and services with substitutes;
- puts a premium on trust-building, clear communication, transparency and a level playing field between interest groups;
- looks to the future, with a premium on inter-generational equity;
- proceeds through experimentation, learning and adaptation, but reflects a precautionary stance, given the many uncertainties; and
- above all, matches policies, rules and targets with the capacities to achieve them.

C&I number amongst several tools to make the transition to such a form of 'sustainable' forestry. In this section, I concentrate on landscape and national level C&I, because they encompass the critical social and political dimensions of forestry. Ruitenbeek and Carter (1998) caution that '99 per cent of forestry literature focuses on the stand level'; yet 'even with all the tools of good forest stand management at our fingertips, circumstances may go terribly awry if we neglect... the policies and institutions [that govern forest use.]'.

4.1 C&I as tools of compromise

C&I can provide specificity about the forest goods and services which are required, whilst allowing local interpretation and target-setting. C&I potentially provide a language where very different objectives can be brought together, and integrated, traded-off or modified in light of real local needs and capacities. They are therefore tools of *compromise* – and the ability to make compromises, as Edmund Burke postulated, is the foundation of all progress.

The problem here is that C&I may already have a bad image. They are often taken to describe Utopia, a state which accommodates neither needs nor compromise. C&I processes are accused of being both perfectionist and overly comprehensive: ultimate sources of 'policy inflation'. In practice, C&I will work where they enable us to make strategic and focused choices. These should be based on comprehensive *understanding* of the complexities, but not necessarily force comprehensive *action* in every circumstance. For example, as we have pointed out, the best approach to conserving biodiversity might not be to demand it in every forest stand, but to encourage its maintenance at the socially most relevant level – perhaps as a mosaic at the landscape level (allowing plantations to get on with the job of growing fibre).

4.2 C&I as an interface between science and social processes

Reaching compromise is fundamentally a social or political process. Hence *participation* is essential in setting C&I for good forestry, and particularly at the landscape and national levels. It may be worth reflecting upon the recent experience of how various sets of C&I have evolved. Who was behind the process? Who contributed, and who did not? Who resisted or dropped out? How was the process organized, and how was it resolved?

We can learn from International Standards Organization's (ISO's) routine, multi-stakeholder approach to setting international standards used in many types of industry. The Forest Stewardship Council (FSC) followed the general ISO approach in negotiating its principles and criteria. However, aware that a broader scope of participation is required for forestry than is normal for ISO standards (given the extensive externalities), the FSC formed economic, social and environmental chambers of interest groups to generate and keep the principles and criteria under review. However, it may prove counterproductive in the long term to keep people in such 'fixed' groups – especially if the aim is to get all interests to consider economic, social and environmental goals in a balanced way.

There is some validity in the observation that recent sets of forestry C&I tend to have been 'quick fixes' and centralized solutions, and thereby may be predisposed to top-down control and implementation, whether intended or not. They might be contrasted with, e.g. C&I for organic agriculture, which have been slower to evolve (over 50 years, in fact) and based on mutual recognition of many local standards.

The continuing challenges of participation in C&I will not be met overnight. It is certainly not a question of 'policy by brainstorm or phone-in'. There are underlying philosophical arguments which need to be explored and resolved, processes which will neither play out quickly nor be confined to the forest sector. If language constrains agreement, this is often because the underlying philosophies both clash and are unexplored. The notion that forestry goals (and hence C&I) should accommodate *efficiency* is widely accepted. The goal of *equity* is gaining ground. But a conundrum is presented by the emergence of *sustainability* as a goal:

- do we consider sustainability to be nothing more than a *technical constraint* to developmental goals, related to environmental limits – with the implication that this is primarily a matter of science? (Marcuse, 1998); or
- do we consider sustainability, like liberty or justice, to be a *social goal* to which we aspire – but which has to be articulated and agreed locally before it can be achieved in practice – and therefore is primarily a matter of participation (Holmberg *et al.*, 1991)?[1]

If the second is the case, then the predominant political culture will fundamentally condition the interpretation of 'sustainability'. Schanz (1998) lists a range of possible interpretations, in brief:

- in a hierarchical culture, sustainability is about control systems over large spatial scales;
- in an egalitarian culture, sustainability is about responsibility and coordination;
- in an individualistic culture, sustainability is about facilitation and incentives; or
- in a fatalistic culture, sustainability is about reacting to events.

Even if we believe that sustainability is primarily a socio-political construct rather than a scientific concept, there are *clear roles for scientists, social scientists* and *economists* in proposing C&I for good forest management. Science should play its part in accurately describing the physical, social, environmental and economic C&I of good forestry, especially at the stand level. This means that C&I language should be accessible to all, the dimensions that are being considered reflecting interest groups' concerns. Otherwise 'scientific' C&I, too, could serve to increase top-down control.

4.3 Practical opportunities to use C&I to achieve security of forest goods and services

We have said that much effort has been expended in defining C&I for forestry. This does not mean that this work is all over. C&I are only hypotheses at this stage, and need to be put to practical use and tested. Clearly we should be integrating C&I into foresters' management systems. In addition, I conclude this chapter by sketching out a number of highly significant initiatives which we should be seeking to influence:

1. The *Intergovernmental Forum on Forests* seems (rightly) to be intent on avoiding international ‘acronym fever’ – the imposition of precepts such as the previous Tropical Forest Action Plan – and instead emphasizes agreed C&I that are employed through local participatory processes. This provides a key political opening for interest groups and scientists to work together.
2. The *World Trade Organizations (WTO’s) Technical Barriers to Trade (TBT)* principles need reviewing, i.e. what criteria are acceptable in order to protect a nation’s environment and the health of its people? Sooner rather than later, cases will be heard by the relevant WTO panel to see how far certification is a legitimate TBT. C&I should be fed into the panel’s decisions, and so help the TBT agreement to evolve.
3. There is a largely parallel evolution of *C&I for sustainable development*. The world has moved on from describing the success of nations in terms of gross domestic product alone, or in social terms, e.g. the United Nations Human Development Index. Currently the International Union for the Conservation of Nature (IUCN) is working on a national ‘sustainability barometer’ which assesses the direction of change on two main axes: human well-being and ecosystem well-being. There is scope for forest quality criteria, and criteria of forest security, to be included in national ‘barometers’.
4. The *finance and insurance sectors* are becoming increasingly aware of environmental and social risks in forest activities, and are beginning to accept the long-term value of SFM operations as opposed to asset-stripping operations (partly because of the development of new markets through certification, ecotourism, carbon offsets, etc.). No C&I are, as yet, routinely employed in these important decision-making areas.
5. *Local government and Local Agenda 21* initiatives are beginning to take control of certain aspects of forestry. This reflects both a global trend towards decentralization, and the success of Local Agenda 21 in many countries in consulting local groups on forest and landscape values. Ways should be sought to define what forest criteria really count locally: participatory appraisal and monitoring methodologies might be employed to turn the C&I process ‘on its head’. C&I could thus be described in ways which make sense to ordinary people (and not just men in white coats).
6. The corporate sector is rapidly developing *environmental/quality management systems for good forestry*. The ISO forestry working group has prepared a document describing forestry C&I and standards that might inspire companies as they set their EMS targets. Leading corporations can apply tremendous creativity and resources to the practical task of commercial SFM, once societally agreed C&I are made clear and are reflected in market or policy demands (IIED, 1996).
7. *Ecolabel* initiatives are evolving, notably in Europe. Here, the challenge is to encourage the analogous development of C&I in other sectors, so that *substitutes for forest goods* can be treated comparably by consumers and authorities.

8. The further development of options for *international legal instruments concerning forests* is a key element of IFF's work plan. Whether this is a specific forest convention, or the better exercising of existing forest-related conventions such as the Biodiversity Convention or Climate Change Convention (below), C&I will be central. Indeed, it might be argued that an agreed common global set of C&I could fulfil many of the international requirements that are not yet satisfied, thereby obviating the need for further global legislation.
9. A particularly urgent case in which forestry C&I need development and promotion is *carbon offset forests*, and the implementation of the Kyoto Protocol. This issue is worth exploring in a little more detail (Box 3.2).

Box 3.2. Carbon offset forests: the need for C&I.

The Kyoto Protocol, under the Framework Convention on Climate Change, has treated only a part of the carbon cycle: not unnaturally, parties were concerned about burning fossil fuels that have taken geological time to develop. However, there are other segments of the carbon cycle to be dealt with, notably carbon sinks and the sequestration services of forests. Perhaps less attention was given to these because intergovernmental forest agreements have been weak – no 'steer' was given on how good local forestry can contribute to carbon storage. There were also uncertainties about the relative strengths of different forest (management) types as sinks – which meant that Kyoto's number-crunching sessions, rushing to develop formulae, avoided the difficult equations surrounding sinks.

There is a real danger that the Kyoto Protocol and the associated Clean Development Mechanism – if they yield to pressures to handle sinks – will be able to deal only with simple forestry models. Afforestation/reforestation and/or the set-aside of protected areas provide big, easy-to-measure blocks. Furthermore, the development banks see themselves as catalysts for greatly increased private sector investment in carbon offset forests (possibly through stock/commodity markets). The question is where this will help the security of forest goods and services sought by local interest groups, and where will it result in corporations capturing even more value and land. Will it lead to pressure for deforestation followed by subsidized afforestation; and will it push out natural forest management, rotational shifting cultivation and farm forestry?

Certainly, international carbon offset protocols could have enormous implications for:

- the siting of forests;
- the composition of forests;
- 'permissible' activity in forests;
- who gets forest and carbon benefits; and
- who is effectively in charge of forestry.

The reality of C&I for good local forestry needs to be introduced into the carbon equation, so that it can be part of what nations 'sell' on the emerging world carbon offset market. Trees on farms, agroforestry, shifting cultivation and natural forest management may be difficult to measure in climate terms, but they sustain livelihoods and other forest benefits. Managed forests such as these could also store/sequester more carbon than simpler plantation or set-aside systems.

The irony is that it is now well understood that the development of new global markets for forest goods and services should include social and environmental externalities. The danger is that they will be forgotten in the rush to develop the carbon market.

A similar case can be drawn for global biodiversity payments under the Biodiversity Convention. The language of good forest use to secure desired goods and services needs to be introduced here, too – C&I as the 'words' and bottom-up national forest programmes as the 'syntax'.

Notes

1 Taking the justice analogy further, perhaps we should be using C&I to assess what is *not* sustainable, rather than what *is* sustainable; like the justice system, it is far easier in forestry to demonstrate what has gone wrong rather than what is going right.

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4

Between Voodoo Science and Adaptive Management: the Role and Research Needs for Indicators of Sustainable Forest Management

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The development of appropriate indicators of sustainable forest management is attracting considerable interest, although some controversy still exists regarding their utility. The 'wrong' indicators can lead to negative, destructive effects on the systems whose maintenance they allegedly seek to support, and in order for indicators to fulfil a legitimate role as information tools for forest managers, attention needs to be paid to their proper selection, development and use. This chapter examines how these conditions can be met.

As all definitions of sustainability involve value judgements we advise caution when establishing 'objective' and scientifically based indicators, if the assumption thereby is that such indicators might deliver value-neutral information. Attention therefore must be paid to underlying mental models and subjectivity. If indicator sets are to fulfil their goal they must be both transparent and accepted by society.

A hierarchical approach remains currently the most practical and effective model for dealing with criteria and indicators (C&I). C&I sets might, however, be better considered as information or communication networks; as such, C&I would allow us to recognize information related to systemic sustainability.

Thresholds for individual indicators are important as they could theoretically indicate switch points or inflection zones for the system, including points at which the system degrades irretrievably. One major

challenge facing researchers is the identification and quantification of such thresholds. Adaptive co-management is emerging as a promising means of resource management, but there is little understanding of how to maximize its contribution to sustainable tropical forest management. Our vision is that C&I would eventually become an integral part of the monitoring and feedback systems, thus facilitating adaptive co-management.

We conclude that, while indicators suffer from limitations in providing broad but not deep insights, their advantages as effective information tools in the service of forest management outweigh these limitations. They are transparent and readily understandable information tools and their scope is limited only by the extent of our knowledge of the underlying systems and the translation of that knowledge into effective indicators.

1 The cold shadow world of indicators?

The use of sustainability indicators is not without controversy. Bradbury (1996), for instance, argues that indicators are wrong because 'they are a pathological corruption of the reductionist approach to science'. He makes the point that reductionist approaches are incapable of dealing meaningfully with complex systems, and therefore any excessively reductionist approach, such as indicators, would result only in a caricature.

Let us stay with complex systems for a while. A forest ecosystem, coupled in turn to a variety of political, institutional and social structures, is an example of a complex system. It is capable of self-organization in response to environmental changes such as random external shocks. The complexity paradigm has brought with it a deeper and better understanding of irreversible versus reversible and stochastic versus deterministic phenomena. One lesson from complexity theory is that uncertainty is much more pervasive than was ever previously imagined. Ruitenbeek and Cartier (1998) suggest that small initial shocks can, through various feedback mechanisms, have substantial impacts. Arbitrary factors such as political corruption, war, social unrest or changes in international markets can all undermine even the best laid plans for sustainable forest management. For decision makers, addressing such uncertainty is a challenge. As Sterman (1994) points out 'the heuristics we use to judge causal relations lead systematically to cognitive maps that ignore feedbacks, multiple interconnections, non-linearities, time delays and the other elements of dynamic complexity'. This leads to difficulties, as cause and effect are often distant in time and space, and the delayed and distant consequences are often different from, and less salient than, proximate effects, if they are known at all. We are thus confronted with the twin problems of causal complexity and our own cognitive limits to grasp complexity.

All this seems to militate against a role for indicators, suggesting that Bradbury (1996) was right when he exhorted a departure from the 'cold shadow world of indicators' likening their development to voodoo science. Certainly there is sufficient evidence that the 'wrong' indicators can lead to negative, destructive effects on the systems whose maintenance they allegedly seek to support (Fearnside, 1989; Peet *et al.*, 1996; Vanclay, 1996).

In this chapter we will argue that indicators need not be reductionist, indeed they can be holistic and pluralistic and a whole system of indicators can be greater than the sum of its parts. We will discuss their role as communication devices, arranged in a network involving complex interactions and interrelationships, synthesizing many subjective perspectives. We will also argue that research to develop such indicator systems must be pluralistic and interdisciplinary, not positivist or reductionist. This research is necessary because indicators can effectively provide the feedback and communication we need to manage complexity.

2 Terms and conditions

Before we begin our discussion of C&I we need to define what we mean by the terms. After Maini (1993, 1996), Brand (1996) and Prabhu *et al.* (1996), C&I are the tools which can be used to conceptualize, evaluate, implement and communicate sustainable forest management. They are thus essentially information tools in the service of forest management.

Prabhu *et al.* (1996) have distinguished between assessment and monitoring, largely to account for the difference between the use of indicators by external assessors and their use as an integral part of a management system. They have also stressed the difference between both of these terms and guidelines. An important, but often underestimated, function of indicators is their use as communication tools to facilitate consensus building around a common conceptualization of sustainable forest management. For instance, indicators have catalysed the process of developing Regional Forestry Agreements in Australia, they have had a similar effect in Canada through the work of the Canadian Council of Forest Ministers (CCFM, 1997). The Forest Stewardship Council (FSC)'s extensive consultative process on the development of their principles and criteria is another case in point (Erwin, 1996). The African Timber Organization's comprehensive programme on testing and developing C&I is largely fuelled by the desire to develop a common operational conceptualization of sustainable forest management among their member countries, in the expectation that this will result in better regional forest policies and management systems (Bouvard, 1998).

For the purposes of this chapter we will use the term 'indicators' in a general manner to include criteria as well, and will follow Prabhu *et al.* (1999a) in defining them:

Criterion: *A standard that a thing is judged by*. Criteria are the intermediate points to which the information provided by indicators can be integrated and where an interpretable assessment crystallizes. Principles form the final point of integration. Criteria should be treated as reflections of knowledge, where knowledge is the accumulation of related information over a long period of time. It can be viewed as a large-scale selective combination or union of related pieces of information.

Indicator: An indicator is any variable or component of the forest ecosystem or the relevant management systems used to infer attributes of the sustainability of the resource and its utilization. Indicators should convey a 'single meaningful message'. This 'single message' is termed information. It represents an aggregate of one or more data elements with certain established relationships.

It is fair to say that the development of indicators received an enormous boost following the United Nations Conference on Environment and Development (UNCED) in 1992. However, indicators had been used in forest management for a long time before then (Speidel, 1984). It was with the development of the International Tropical Timber Organization (ITTO) C&I that the current wave of indicator development for tropical forest management began (ITTO, 1992). Granholm *et al.* (1996), Prabhu and Tan (1996) and Grayson and Maynard (1997) provide an overview of the various initiatives ('processes'). There is an extensive literature on environmental indicators (McKenzie *et al.* 1992; Bakkes *et al.*, 1994) and sustainable development indicators (Organization for Economic Cooperation and Development (OECD), 1993; Scientific Committee on Problems of the Environment (SCOPE), 1995; World Resources Institute (WRI), 1995), which will not be reviewed here. We will also not review alternative approaches to assessing the state of natural systems, such as the AMOEBA approach (Brink *et al.*, 1991), the 'barometer' approach (Prescott-Allen, 1995), corporate environmental performance reporting (Cook and Stevens, 1992; Ditz and Ranganathan, 1996), etc.

In our view the utility of indicators as information tools is contingent on the satisfaction of the following three conditions:

1. The selected indicators are relevant to the assessment or monitoring goal, in our case sustainable forest management.
2. In its entirety the set of indicators is sufficient to deliver meaningful information about the development trends in the underlying ecological and social systems, and are useful to the determination of policy/management responses.
3. Non-linear and compensatory effects among indicators are adequately understood.

In addition to these three conditions Prabhu *et al.* (1996) list seven other attributes that can be used to improve the quality of indicators.[1] In the following we present our ideas on how to meet the three conditions listed above.

3 Value judgements and objectivity

In its essence sustainability is the ability to maintain something over time. While several definitions of sustainability exist (e.g. ITTO, 1991; Ministerial Conference on the Protection of Forests in Europe, 1993), it is important to stress that all these definitions involve value judgements, are based on some amount of consensus and reflect particular world views. For instance these definitions might suggest the maintenance of well-being, material and spiritual, of all human beings, but may vary in the importance they accord small communities versus the common good,[2] or may stress the well-being of the current generation over the future. All these are value judgements. As Lélé and Norgaard (1996) point out, subjectivity is not a phenomenon of the social sciences alone; biophysical sciences are also prone to making value judgements. This takes place for instance when the framework for analysis or model construction is chosen. Different frameworks stress different factors, pay less attention to others, and ignore most. In 1989, proponents and opponents of forestry in the tropical rainforests of the Atherton Tablelands, in far north Queensland, deposited arguments for their respective cases as sworn affidavits in a court of law (e.g. Ashton, 1989; Nicholson and Keys, 1989; Keto *et al.*, 1990; Vanclay *et al.*, 1991; Vanclay, 1992). Arrayed against each other were some of the foremost forest ecologists of the time. The substance of the ecological arguments can be summed up as a clash between those who believed that tropical rainforest dynamics were best captured by equilibrium models, and those who held a non-equilibrium view. Although the decision to declare the forests a World Heritage Area was ultimately taken in the political arena, this is a good example of how belief systems can affect the evaluation of biophysical sustainability.

We therefore advise caution when the call is given to establish objective and scientific indicators, if the understanding thereby is that such indicators might deliver value-neutral information. Guha (1985) has pointed out that the history of tropical forest research shows a strong correlation between what is considered essential by the resource owners and the research carried out by scientists. This nexus has historically ignored the needs of actual resource users. We therefore agree with Lélé and Norgaard (1996) that scientists should accept the inevitability of making value judgements in the process of their research, and should make these value judgements explicit to those most likely to be affected by them.

Consequently, for research on indicators it is important to pay attention to the need to bring to the surface underlying mental models and subjectivity, and to communicate these effectively. If indicator sets are to fulfil their goal they must be both transparent and accepted by society. However, it is not always clear to the developers and users of indicators themselves what belief and value systems are driving their quest for information. For instance, indicators of 'social change' such as gross domestic product (GDP), financial market indices, unemployment, etc., have been criticized by Peet *et al.* (1996) as being better suited to maintaining the *status quo* rather than to promoting change. Research could make an important contribution by exposing the underlying assumptions in sets of indicators, challenging these assumptions where necessary, thereby catalysing improvements to the nature and composition of the indicators used. Examples of the influence of value judgements from the realm of sustainable forest management indicators include the use of keystone species, especially in tropical forests (Landres *et al.*, 1988; Stork *et al.*, 1997; Lawton *et al.*, 1998) and the concept of 'authenticity' as proposed by the World Wide Fund for Nature (Denevan, 1992; WWF, 1993).

Thus, in selecting sustainability indicators we need to be clear about whose definition of sustainability we accept as the framework. This definition will then need to be deconstructed into information targets, and the time-horizons over which they apply. The trade-offs between indicators will have to be resolved and a social consensus amongst those affected attained. Only when clarity with respect to these issues exists can real progress be made. Some examples of frameworks for indicator development include Smyth and Dumanski (1993), International Union for the Conservation of Nature (IUCN, 1995), Dykstra and Heinrich (1996), Borrini-Feyerabend (1997), Stork *et al.* (1997), Center for International Forestry Research (CIFOR, 1999) etc., along with the other better known processes.

4 Hierarchies and networks: a systemic view

Most sets of indicators currently in use or under development are represented as hierarchies of concepts (ITTO, 1992; Amazon Cooperation Treaty, 1995; CCFM, 1995; Prabhu *et al.*, 1996, 1999a). In these hierarchies indicator clusters are subordinated to criteria. This hierarchical relationship might then be extended in either direction to form a tree. Such hierarchies facilitate organization and integration of information. They also facilitate communication about the model of sustainable forest management to be attained. A precondition to the development of a hierarchy is the determination of specific links and relationships among the C&I. A comparison of the various sets of C&I currently in existence will reveal that there is no single common hierarchy in use across these sets (e.g. Lammerts van Bueren and Blom, 1997). Often the same or similar elements are treated very differently from one set of C&I to the other. However, if the value of information lies in the way it is organized (Larsen in Rauscher and Hacker, 1989), then it would be important to try and resolve these discrepancies.

Analysing some of the confusion in the use of terms and the relationships between hierarchical elements such as principles, criteria and indicators, Prabhu *et al.* (1996) adopted a four-step hierarchy, with specific functional relationships between hierarchical limits. In later work, Prabhu *et al.* (1999a) have linked these concepts to the four basic entities in information theory proposed by Liang (1994), thus opening a conceptual link between C&I and information management. To date the most thorough treatment of indicator sets as hierarchical models has come from Lammerts van Bueren and Blom (1997), who have devised simple rule sets to facilitate operational use of the hierarchical approach. The hierarchical approach remains currently the most practical and effective model for dealing with C&I. A more widespread and rigorous use of this approach would reduce some of the redundancy and noise that currently besets most sets of C&I. Use of such an approach would also facilitate comparison of indicator sets between regions and sectors.

However, as Prabhu *et al*. (1996) and Burford de Oliveira (1999) indicate, some of the problems encountered may be systemic and a result of constraining the representation of indicator sets to two-dimensional hierarchies, instead of recognizing the four dimensions of time and space they actually occupy. Instead of hierarchies, C&I sets might be better modelled as information networks. We follow Fink and Kaplowitz (1993) and describe a network as a set of linkages between various entities that allows the transmission of matter, energy or information. In our case the entities are C&I and the focus is on information flux. In our understanding of how network models work we follow Barnett and Rice (1985) in viewing information as flowing between nodes; these nodes might consist of individual elements, such as indicators, or of groups of elements. Woelfel (1993) defines communication networks as sets of nodes whose state is at least partly a function of the states of other nodes in the set. He goes on to define cognitive processes as the changing patterns of activation of nodes in a network, and cognition as being an emergent property of an underlying communication network. This reinforces our belief in the importance of identifying nodes and nodal interactions, reducing the focus on the individual indicator. It is our understanding that C&I represent a form of communication network, with the special utility of the C&I network being to facilitate cognition of the state of sustainability in the forest–human systems in question.

In other words, when modelled as communication networks, C&I allow us to recognize information as patterns of nodal activation values in response to external stimuli, i.e. changes in the underlying ecological and social systems. This kind of information relates to systemic sustainability, whereas the activation state of a single indicator may or may not be relevant to sustainability, depending for instance on whether compensatory effects are taking place. Thus, interpretation of such nodal activation patterns would be critical to the understanding of the dynamics of the underlying systems. To our knowledge, so far very little research has focused on identifying the information contained in such activation patterns. Such research would allow us to better understand non-linear and compensatory effects among indicators, for instance by comparing indicator linkages in critical versus non-critical patterns. As critical information, we see nodal activation patterns that provide information directly related to the three central questions of sustainability postulated by Peet *et al.* (1996):

1. Are all of the people well-off, satisfied, happy?
2. Are we achieving the maximum possible well-being with the least possible throughput of material and energy?
3. Are the natural systems that supply resources and accept wastes healthy, resilient, and full of evolutionary potential?

If C&I systems can be viewed as a communication network acting as a parallel distributed processing (PDP) system,[3] as we believe they can, then Richards (1993) offers some insights into possible research strategies. From a different perspective, research on exploring linkages and testing sets of 'sustainability' indicators by Lonergan *et al.* (1996) supports the view that efforts to focus on precise movements of specific indicators are misplaced since, in reality, most policy choices are themselves relatively imprecise. Practical and immediately relevant research on recognition of critical nodal activation patterns is being proposed by Ghazoul and co-workers (London, 1998, personal communication) in the field of biodiversity indicators through recognition of the need to work on the interlinkages among indicators.

Very little experience exists on the practical benefits of modelling C&I sets as networks. Prabhu and Sukadri (unpublished data) have modelled results from CIFOR's research on C&I as semantic networks using SemNet,[4] in an effort to better understand information linkages between the elements of the C&I hierarchy. Based on this early understanding of relationships between the hierarchical elements, researchers at CIFOR are seeking to model C&I as hierarchically defined objects capable of cross-linkages, information sharing and evolution, whereby evolutionary pressure is exerted by human users of the computer system within which the objects reside. The application for which this network is being developed is a practical training and development tool (Haggith *et al.*, 1998; Prabhu *et al.*, 1999b), but the pursuit of more ambitious research goals such as the study of system structure and behaviour could also be envisaged using similar techniques. In a more pragmatic use of tools based on network theory, Colfer *et al.* (1996) have used the GALILEO and CatPac software packages to develop and research C&I related to social sustainability (Woelfel and Fink, 1980; Terra Research and Computing, 1990, 1995).

5 Towards a structured selection of indicators: identifying the common ground

Having identified a conceptual framework, how does one identify a set of indicators? Given the large number of C&I sets under development, the easy way would be simply to adopt an existing set. Quite apart from any other inconsistencies, this could easily result in an unwitting adoption of value systems and goals that do not enjoy a social consensus in the context where they are to be used. This may be especially inappropriate if these sets of C&I have been developed in quite different social and ecological contexts.

Most definitions of sustainability were previously seen to converge on the three central questions identified by Peet *et al*. (1996). This is a good first step in our search for an appropriate set of locally adapted indicators. Obviously these are very broad questions, and therefore interpreting them as indicators will not be easy. The next step might be to consider what Bossel (1996) has called *basic orientors*, the entirely general properties that, he suggests, are important for every system. These properties influence not just the structure and function of the system itself, but also influence that system's behaviour towards its surrounding environment. A system that is equipped for securing better overall orientor satisfaction will have better fitness, so evaluation of orientor satisfaction is therefore a means of measuring system fitness. Bossel suggests this can be done by means of indicators. He postulated six basic orientors and three additional basic orientors for situations involving living creatures (Table 4.1).

Bossel (1996) also mapped the nine basic human needs postulated by Max-Neef (1991, cited in Bossel, 1996) on to seven of the nine system orientors. Earlier we suggested that the first condition for the utility of indicators is their relevance to sustainable forest management. We suggest that Table 4.1 provides a good definition of the term 'relevance' from the broad standpoint of sustainability as it applies to human and living systems in general.

Using a comparative case study approach, Prabhu *et al*. (1996), Cossalter (Bogor, 1998, personal communication) and Burford de Oliveira (1999) have identified indicators of sustainable forest management shared in common between widely differing situations in natural forests, plantation forests and community-managed forests, respectively. Granholm *et al*. (1996) have also identified commonalities amongst several different initiatives on C&I. At the same time these authors have also stressed the differences in the results

Table 4.1. System orientors and human needs.

Bossel's basic system orientors	Max-Neef's human needs
Existence[a]	Subsistence
Effectiveness[a]	Understanding, idleness
Freedom of action[a]	Freedom
Security[a]	Protection
Adaptability[a]	Creation
Coexistence[a]	Participation
Psychological needs[b]	Affection, identity
Reproduction[b]	
Responsibility[b]	

[a]Basic system orientors; [b]additional orientors for living creatures.

obtained from the case studies. This suggests it would be unreasonable to expect the same set of indicators to apply universally, a conclusion echoed by others (e.g. Attiwill, 1996). However, it does seem reasonable to use sets of C&I developed in similar contexts as starting points for the development of locally adapted and appropriate indicator sets.

6 Dealing with uncertainty

Any attempt to develop indicators for sustainable forest management sooner or later faces the need to address uncertainty. From our analysis of the literature we find three basic strategies are used. The first is simply to ignore uncertainty, and therefore to deny complexity. This may be a pragmatic approach to developing indicators if these indicators are to be used only over a very short period, or in other words, if a static view of dynamic systems seemed justified. For instance, some sets of indicators for certification assessments of forest management emphasize particular management prescriptions over the outcomes they cause, thus suggesting that the current 'state of the art' provides the best course. Over time, however, such approaches are bound to restrict innovation and are therefore likely to be counterproductive.

The second potential approach involves trying to decide whether or not complexity is in fact relevant to a given problem, and then either dismissing it or dealing with it. Following early theoretical work by Hammond (1983) and subsequent work by Jack (1993), there are some rule-of-thumb tests in economics that basically permit us to decide whether we are looking at a 'marginal' change or a substantive change. When there are marginal changes, we assume that there is a shift from one partial equilibrium (PE) to another PE and that no bizarre dynamic destabilizing effects take place. A rule of thumb that Ruitenbeek (1994) and Ruitenbeek and Cartier (1995) have successfully used is that if an economic change is more than 10% of a major system variable (employment, GDP, etc.), then it is non-marginal and there is need to be concerned about uncertainty and non-marginal effects.

The third approach is to assume the system is complex and therefore confront uncertainty head on. Ruitenbeek and Cartier (1998) respond to uncertainty by suggesting the inclusion of the 'precautionary principle' into any set of indicators of sustainable forest management. From an economics perspective they also recommend the inclusion of indicators that reflect the need to address efficiency and (intra-generational) equity. They also recommend avoiding (i) the use of internal rate of return, which is a frequently used but inaccurate measure of economic efficiency; (ii) valuation of biodiversity, carbon sequestration and certain ecological functions that are equally well captured by a simple physical accounting of the forest biomass; and (iii) use of complex economic indices and coefficients to characterize income distribution concerns.

Stork *et al*. (1997) developed a framework that allowed them to address both uncertainty and cost in the context of indicators for the conservation of biological diversity in managed production forests in the tropics. Using this framework (Fig. 4.1), the authors postulate that changes in biodiversity may be assessed indirectly through assessment of the processes that maintain and generate biodiversity.

Their expectation is that this approach will allow them to determine how a new level of biodiversity, after a management intervention, would fit with the previous trend, and whether the new level is sufficient to support all-important ecological processes. Indicators may be identified at any point in the causal chain from human intervention to biodiversity. By articulating this systemic view they avoid having to deal with some of the uncertainty related to the definition of biological diversity and the costs associated with taking inventories of species in tropical forests. This approach embodies the notion that the most useful information about a complex system comes from a study of the systems structure, i.e. the structure of biological diversity, followed by its behaviour, which is determined by the processes that shape and maintain biodiversity. On the other hand focusing solely on 'events' resulting from the behaviour, i.e. the mediators, can be quite misleading.

This approach allows us also to understand the degree of surrogacy, i.e. whether and what kind of proxy indicators are in use. Quite early Stocking and Abel (1981) concluded that the interdependence of environmental factors

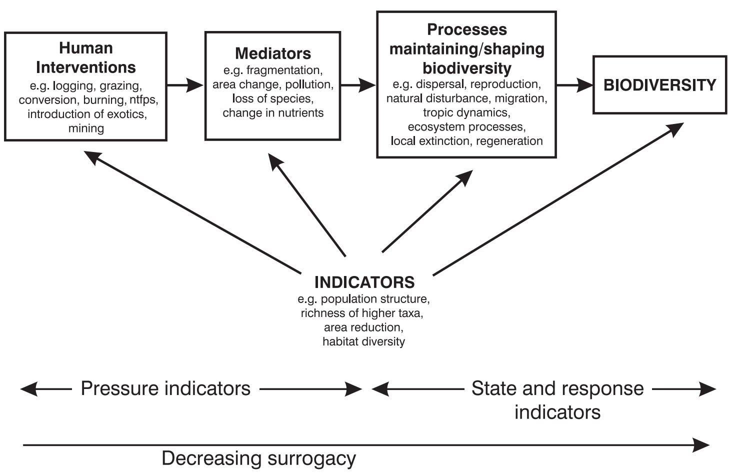

Fig. 4.1. Conceptual framework for the development of biodiversity indicators (after Stork *et al*., 1997). Two steps in the chain of cause and effect are identified: mediators are the immediate physical consequences of each category of human activity; ecological processes respond to the mediators and, in turn, determine the magnitude and maintenance of forest biodiversity. ntfps, non-timber forest products.

afforded opportunities for the definition of proxy measures. Proxy indicators are also used quite frequently when attempting to deal with ‘soft variables’ such as motivation, commitment, fear, happiness, etc. The challenge for research is to attempt either to improve the quality of proxy indicators or reduce the degree of surrogacy while maintaining cost-effectiveness of indicators.

Kimmins (1990) stresses the need to develop a ‘temporal fingerprint’ of forest ecosystem condition based on which trends of degradation and recovery might be assessed. Although theoretically appealing, this idea is difficult to put into practice in humid tropical forests, given the dearth of information on these ecosystems, and uncertainty about what is a reasonable baseline for comparison.

Confronted with the complexity and diversity of social systems, Colfer *et al.* (1998, 1999a,b) have adopted a mixed strategy. They have complemented work on the development of indicators on the one hand with the development of more subjective holistic assessment techniques with which to assess the state of these indicators. The emphasis is very much on iterative assessments and dialogue within a framework of social science indicators.

Lonergan *et al.* (1996) looked at system dynamics as well as indicators in developing their simulation model of complex systems in the Fraser River basin of Canada. They attempted to model ‘critical’ points at which the system may start to collapse because of negative feedback loops. The main ‘feedback’ in that case was migration from the area, in response to things like crime and economic circumstance. It was destabilizing if the migration also took away wealth and a tax base. Ruitenbeek and Cartier (1996) used a similar approach to model nuclear accidents in Vancouver harbour from nuclear submarine testing. These are simulation modelling approaches to uncertainty using dynamic complex system analysis and indicators to determine safe thresholds and ‘precautionary’ policies. We will return to thresholds and the use of indicators in simulation modelling later in this chapter.

7 Inputs or outcomes?

An examination of most C&I sets will reveal that they are a mixture of input-, process- and outcome-based indicators. Input-based indicators focus on human inputs to the management of forest resources; these are tangible and material. Process-based indicators in this context would focus on human management processes, i.e. actions. Outcomes then are the impacts of both inputs and processes. The rationale for the choice of process- or outcome-oriented indicators is usually not obvious from reading these documents; indeed in some cases it is possible that no conscious thought was given to this matter. Yet in terms of the information efficiency and cost-effectiveness of indicators it is important that a conscious choice be made in this regard. After

Meredith (1996), we identify four advantages of outcome-based indicators over input- or human management process-based ones; they:

- require the definition of clear organizational and societal goals;
- allow improved accountability by enabling an improved capacity to judge performance;
- allow and encourage organizational learning and improvement; and
- enable comparisons between organizations.

On the other hand, input- or management-process-based indicators are easy to formulate and easy to measure. They are also easy to communicate and if the intention is to promote a particular set of management interventions, then such process or input indicators will be of advantage. In certification contexts where evaluation of compliance with a management process is important, such process-based indicators are useful. The danger over the long run is that management might become locked into systems that are incapable of adapting to change. If we cluster indicators around broad themes such as ecological integrity, economic viability, social sustainability and production systems, then indicators based on human management processes might most naturally be clustered around production. Outcome-based indicators would better fulfil information needs for the other three thematic areas, i.e. policy, ecology and society.

Analogous to the discussion on input, process and outcome indicators in human systems, Brown *et al*. (1997) suggest that pressure indicators are easier to develop than state or response indicators, but provide much less valuable information. Response indicators, potentially the most valuable indicators, are also the hardest to develop and apply. The terms pressure, state and response indicators are used in the original ecological context (Friend and Rapport, 1979), not in the sense of the OECD framework (OECD, 1993).

It should be noted that social scientists working on the CIFOR C&I project have consistently found it difficult to align their models of human–forest interactions with the frameworks described above. This suggests that some paradigms in science may be more readily able to use these frameworks than others. This serves to illustrate some of the difficulties of interdisciplinary research.

8 Scales and trade-offs

Although in this chapter we focus on the forest management unit level, the systemic processes that determine sustainability operate at various scales, and as Hoekstra *et al*. (1991) have pointed out, processes at one scale are influenced by processes at others. This is as true for social processes as it is for economic or ecological processes. One approach to the question of dealing with scale is to develop different indicators for each scale. This approach has been adopted in many sets of indicators (e.g. FSC, 1994; Hahn-Schilling *et al.*, 1994; CSA, 1996). Stork *et al.* (1997) have suggested for biodiversity indicators that the best approach to addressing scale would be to begin assessments at broader resolutions and move towards finer resolutions, initially seeking out indicators that offered multi-scale information at each step. Certification assessments will conventionally begin at the broadest scale, moving then to some sampling of finer scales (Upton and Bass, 1995). As implied earlier, it is conceivable that a value of one indicator may imply sustainability, but when combined with the value of a second indicator the overall conclusion may be non-sustainability, reflecting interactions among indicators. In response, for biological diversity Boyle (1998), for instance, suggests the need to develop a joint risk function that provides an overall, integrated assessment of conservation of biodiversity and sustainability of forest management operations.

Faith (1995) offers an alternative approach to evaluating biodiversity at the regional scale. Using sensitivity analysis on different weightings of biodiversity criteria, he is able to determine the trade-offs between current forms of land use and alternatives. He sees this approach as an alternative to multicriteria decision aid (MCDA) techniques (e.g. Romero and Rehman, 1987; Petry, 1990; Munda, 1993). However, MCDA techniques remain by far the most common techniques for a structured and formalized approach to determining trade-offs. Particularly the Analytic Hierarchy Process (Saaty, 1995) has been used successfully in forest management contexts (e.g. Kuusipalo and Kangas, 1994) and is an integral part of the certification assessment process designed by the Indonesian Ecolabelling Institute for Indonesia (see Mendoza *et al.*, 1999, for an example of the application of these techniques to C&I). More recent work, such as by Bousquet *et al.* (1993), on modelling multi-agent systems, and Haggith (1996), on a meta-level argumentation framework for reasoning, could provide new insights into the determination of trade-offs (Bousquet *et al.*, 1993) or analysis and support to multi-stakeholder disagreements or argumentation (Haggith, 1996).

9 Thresholds

We view thresholds in two different ways. The first is systemic for which we cite Zadeh (1973): 'As the complexity of a system increases, our ability to make a precise and yet significant statement about its behaviour diminishes until a threshold is reached beyond which precision and significance (or relevance) become almost mutually exclusive characteristics'. One of the implicit assumptions in carrying out research on indicators is that they are not intended to provide precise statements about the behaviour of complex systems. Their intention should be to provide quick and cost-effective information. In our view, indicator systems should stress relevance over precision. Because of the large number of variables involved in such indicator systems any reliable measurement of the value of the information they provide can only be determined through simulation modelling of their performance. An interesting outcome of one of the few integrated attempts of an empirical nature to use simulation modelling to determine thresholds (Lonergan *et al.*, 1996), is that you do not need a lot of indicators to do the actual modelling and to demonstrate the instabilities. They narrowed the indicator set down to 20, and 12 key indicators would have been sufficient to describe future instabilities as well as track historical changes accurately. None the less Zadeh's axiom still applies, warning us of the limitations of the information indicators can generate in the face of complexity.

The second is thresholds for individual indicators, inasmuch as they could theoretically indicate switch points or inflection zones for the system. Such thresholds might indicate that the system, which might be social, economic or ecological, is changing course and adopting a new, not necessarily less sustainable, pathway. A more severe form of switch point or inflection zone might see the system degrading irretrievably. Some research on thresholds of this nature is taking place, but much of what has taken place has not been very conclusive (Cronan and Grigal, 1995 (cited in Boyle, 1998); Bauhus and Khanna, 1999). Boyle (1998) points out that there is some evidence for the existence of fragmentation thresholds, with regard to remnant population size and isolation, below which reductions in diversity are apparently not encountered. He states that the relationship between population size and allelic richness for *Scabiosa columbaria* suggests that remnant populations larger than 200–300 individuals maintain high diversity, while those lower than this do not. For animal-pollinated and dispersed plants, dispersal, gene flow and recruitment will be reduced if interpopulation distances become greater than the home ranges, or gap-crossing ability, of pollinator or disperser guilds (Powell and Powell, 1987).

Possibly one of the biggest challenges facing researchers currently is the identification and quantification of such thresholds. Although some research on thresholds may be better carried out by mono-disciplinary teams of researchers, by its very nature sustainability demands that research on C&I should be multidisciplinary. Janssen and Goldsworthy (1995) emphasize that successful interdisciplinary research requires a shift from the traditional reductionist approach to a more systems approach.

Most C&I research currently underway is multidisciplinary or interdisciplinary in nature. However, more participatory forms of research may be required to ensure that indicators adequately reflect social consensus, thereby improving the probability that their usage enters the mainstream. This may require the application of participatory research techniques (Selener, 1997) such as those used quite successfully in Nepal, for instance. This is certainly true of research carried out by CIFOR (Prabhu, 1997) and elsewhere (Brand and LeClaire, 1994; Ministry of Environment and Energy, 1996; Simula, 1998). Multidisciplinarity does not always entail collaboration between the social and biophysical sciences. Hopmans (1998) has reported on proposed interdisciplinary research on C&I that will be restricted to the biophysical sciences. Much C&I development takes place in the context of research not targeted specifically at C&I, but on aspects of system interactions and dynamics. Examples of such research are provided by Uhl and Viera (1989), Ong *et al.* (1996), Centre for Forest Tree Technology (CFTT, 1997), Woodley and Forbes (1997) and Sist *et al.* (1998).

10 Moving targets, new technologies: dealing with change

We must recognize that as human society develops so will its understanding of what constitutes sustainability. We are thus faced with a moving target. If indicators are to be useful in pinning down sustainability, they must be capable of providing useful information over longer periods of time. If we approach indicators from an information theory viewpoint, then indicators as repositories of information should be sensitive to changes in knowledge, i.e. advances in our understanding of systems, but less sensitive to changes in data structure. This suggests the need to disassociate the indicators from the means of collecting the data, while maintaining their links to progress in scientific knowledge and social debate on sustainability. Indeed, there may be a need to disassociate them from the nature of the data itself, in order to maintain their relevance in the face of rapid technological innovations.

Let us, for instance, explore how an indicator related to changes in forest cover might be supported by data from conventional terrestrial inventories, or data remotely sensed. The implications are for the quality and cost of the data, not for the validity of the indicator itself. Satellite imagery, which typically operates at resolutions between 10 and 100 m, is being intensively investigated for its ability to identify, quantify and track changes to forest resources (e.g. the European Union's TREES project). Preliminary results indicate that, in some cases, data from visible-range and infra-red wavelength detectors (e.g. SPOT, Landsat) can identify forested and non-forested land with reasonable accuracy, though various types of forest/non-forest interfaces cause classification errors (Sader *et al.*, 1990). So far with a few exceptions, e.g. Brondizio *et al.* (1994) and Dennis *et al.* (1998), most examples of the use of satellite-based remote sensing are at broader scales than that of the forest management unit. For instance an indicator on fragmentation was measured at the continental and biogeographic regional scales, based on classified AVHRR satellite data in Australia (Commonwealth of Australia, 1997). The measurement concentrates on the following attributes of spatial patterns:

- 'clumpiness': the number of small, medium and large patches;
- connectivity and isolation: the distance between patches; and

- patch shape: both external boundary complexity and intrusions and islands of non-forest in forest patches.

However, in other regions, especially in the equatorial zone, where cloud cover is frequent and extensive, visible wavelength detectors are of limited value. The use of microwave (radar) detectors (e.g. on the ERS series of satellites) offers a promising alternative. A possible additional advantage of radar remote sensing is that radar can penetrate not only clouds, but also forest canopies, and even soil. Different radar bands, varying in frequency, wavelength, and/or polarization, have different properties in terms of penetrability and detection capability (Hoekman, 1987). Much work has been done already on the capacity of airborne radar remote sensing for the estimation of forest biomass. These results suggest that airborne radar remote sensing has the potential, at a satisfactory level of resolution, to detect variation in tree height (i.e. the difference between crown surface elevation and ground surface elevation – not detectable by visible wavelength detectors), crown biomass and ground vegetation structure.

As both radar and visible wavelength detectors can be operated simultaneously, the possibility exists for synergism in the use of both (Leckie, 1990a,b). Visible wavelength detectors, as well as near infra-red, can be used to detect variability in crown surface colour and texture, which is also expected to be related to forest structural diversity. Once some of the significant technical problems have been overcome, such as geometric correction, calibration and registration to ground control points, there is great potential for these methods to be used in monitoring indicators. This capacity to monitor changes in diversity over time constitutes a valuable tool for determining the sustainability of forest management practices, and is possibly the simplest and most useful application of airborne remote sensing.

In the social sciences; total forest valuation techniques have been increasingly used recently; the basic approach was developed earlier (Hufschmidt, 1983) but has not yet contributed much to the development of indicators through delivery of data. The same might be said about approaches to biodiversity valuation (Barbier *et al.*, 1994) or benefit transfer work (e.g. Manoka, 2000).

The complex interaction of many factors and the need to forecast possible scenarios make systems modelling a logical way to build on and extrapolate the C&I work. Ongoing modelling work by Vanclay and co-workers on the human–forest interface (FLORES: Forest Land Oriented Resource Envisioning System) should provide interesting feedback to the development of the indicators (variables) modelled (Vanclay, 1997). While C&I offer a better understanding of the many issues impinging on forest management, they do not, in themselves, provide a basis for exploring policy options and predicting future scenarios. FLORES extends the C&I work to provide an objective basis to explore options and implications. Models such as FLORES offer new insights and pose new problems for research.

11 Indicators in the real world: adaptive management, sustainability and tropical forests

The user group for indicators is broad and includes forest managers, certification bodies, policy makers, technical cooperation agencies, scientists and others. We believe that the biggest potential use for indicators for sustainable forest management is in the context of strengthening feedback loops in adaptive management systems. Before we turn to the use of indicators in adaptive management, let us briefly examine the concept of adaptive management.

In terms of lessons for sustainability, Nicolis and Prigogine (1989) note '. . . the adaptive possibility of societies is the main source allowing them to survive in the long term, to innovate of themselves, and to produce originality'. Illustrating complex behaviour using dynamic modelling, these authors demonstrate that a new human activity launched at a certain time can grow and stabilize. Under some conditions it may even compete successfully against similar activities in the course of time. However, this is not a guarantee of success, as the same activity launched at a different time may result in a very different outcome, even in a total loss. This they suggest illustrates the 'dangers of short-term, narrow planning based on the direct extrapolation of past experience'. It was based on a similar understanding of complex systems that prompted Holling (1978) to develop the concept of adaptive management of ecosystems. Adaptive refers to a process of progressive revision involving conscious responsive action to incorporate new knowledge. Adaptive management seeks to get away from the 'command and control' mentality of other management models (Holling and Meffe, 1996). *Adaptive management seeks progressive improvement, by treating each management intervention as an experiment, monitoring its development and learning from the outcome*. This concept has since been applied in a practical context with mostly positive results in a series of 'experiments' in temperate and boreal regions (Lee, 1993; Stankey and Shindler, 1997).

A key aspect of adaptive management is the mechanism by which managers can monitor the outcomes of their interventions and so enable institutional learning. Our vision is that C&I would eventually become an integral part of the monitoring and feedback systems of forest management units.

In the context of tropical and sub-tropical forest areas, the complexity of social systems, and the overlapping of rights, ownerships and tenure, call for co-management of the resources. Co-management involves the collaborative management of a resource by relevant stakeholders whose rights and responsibilities are delineated and shared through a purposive negotiation process. We believe that in such tropical forest areas it is more appropriate to speak of the need for 'adaptive co-management'.

Adaptive co-management, in different forms, has been emerging as a promising means of resource management with forest-related, agricultural and fisheries applications. So far, progress has been via piecemeal and disconnected initiatives, and there is little understanding of how to maximize its contribution to sustainable tropical forest management. There is a pressing need for monitoring arrangements that deliver comprehensive, relevant, scientifically sound and cost-effective information regarding the sustainability of resource use to collaborating actors in forest management. Such monitoring arrangements would take into account the technical 'non-human' dimensions as well as the social dimensions of monitoring, such as what is monitored, for whom, by whom, for what purpose and at what price. This requires a system in which there is little noise or redundancy in the network of indicators, an area for research. Such systems hold great potential to improve the management of complex human–forest systems, through strengthening feedback loops in dynamic management models that satisfactorily support the goals of equitable and sustainable forest management.

12 Conclusions

Fisher (1996) is concerned that indicators might become ends in themselves and warns that there is a tension between the legitimacy and accountability of decision making and its efficiency and effectiveness. Whereas indicators might serve to improve the efficiency and effectiveness of decision making, they should not be the sole grounds upon which decisions are made, as they will not capture the whole reality of sustainability, but only those parts we can measure and of which we have some understanding. Fisher emphasizes, and we agree, there is need to come to a realistic understanding of the role of indicators in decision making related to sustainable forest management.

We have argued throughout this chapter against excessively reductionist approaches to developing indicators, which might cause the kind of concern that Bradbury (1996) justifiably expresses. Instead we have suggested the need to look for holistic indicator systems and the need to understand how information flows across these systems. We have also pointed out the need to recognize the role of belief and value systems in the shaping of indicators, and the importance of being sensitive to complexity and uncertainty. Indicators must be designed to adapt to change; only then will they continue to be useful over the long term. For this their main focus will need to be on system structure and behaviour, not on the pressure we human beings wilfully exert on systems through management interventions.

Simulation modelling may be a better alternative to the use of indicator systems, where appropriate models, data, information and infrastructural support exist, but for most tropical forest countries the extensive use of modelling in decision making is not yet a realistic scenario. Additionally modelling can often be a 'black-box' to outsiders involved in the decision process. Indicators have already been successful in facilitating better communication among stakeholders on sustainable forest management. We believe they also offer real opportunities to improve feedback to decision making and planning. We agree therefore with Margules and Lindenmayer (1996) that '(m)onitoring must not continue to be regarded as second rate science and starved of resources'.

Indicator systems must be both *pragmatic*, in that they bear relevance to management needs and societal expectations, and *realistic*, in that their role as agents for improvement is well defined and their limitations known. Some of these limitations are that indicators will provide broad but not deep insights into situations. The information they provide will seldom be very precise. If we accept these limitations, there is no real reason why indicators should be excessively reductionist or incapable of dealing with complexity. On the contrary, they offer a transparent and readily understandable means of gathering information on sustainability trends and the impacts of human interventions. Their scope is currently limited only by the extent of our knowledge of the underlying systems and the translation of that knowledge into effective indicators.

Notes

1 These are (i) precision of definition; (ii) diagnostic specificity; (iii) sensitivity to change or stress; (iv) ease of detection, recording and interpretation; (v) ability to summarize or integrate information; (vi) reliability; and (vii) appeal to users.

2 Bossel (1996) suggests the need to recognize the ethic of partnership for a viable approach to sustainability: 'All systems that are sufficiently unique and irreplaceable have an equal right to present and future existence and development'.

3 In a PDP the information processing and communication activities are distributed over the entire system, and the elements or nodes act more-or-less in parallel. In the hierarchical network model of C&I we are proposing there is obviously a fair amount of sequential activity as well as parallel activity.

4 See URL http://www.biologylessons.sdsu.edu/about/semnetdown/html for information on the software.

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5

'Whose Forest is this, Anyway?' Criteria and Indicators on Access to Resources

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This chapter provides a brief history of the social component of Center for International Forestry Research (CIFOR)'s project, 'Assessing Sustainable Forest Management: Testing Criteria and Indicators', with special reference to the issue of security of inter-generational access to resources (SIAR). We first present an overview of the literature on assessing SIAR; then we present our social criteria and indicators (C&I) 'best bets' and some methods for assessing them. We conclude with our earliest tentative findings relating to the possible causal links between sustainable forest management on the one hand, and one element in SIAR (sharing of forest benefits), on the other. In this analysis we used data from two forest-rich sites (Bulungan, East Kalimantan, Indonesia and the Dja Reserve, Cameroon) and two forest-poor sites (Long Segar, East Kalimantan and Mbalmayo, Cameroon). We found that timber companies and the government were perceived to have the dominant shares of cash and timber in all sites, though to varying degrees; and that local communities were seen to have dominant shares of other forest products. Differences based on forest quality were not striking. We conclude by discussing our plans and recommendations for future research.

1 Research approach

One may first ask why we care whose forest is being managed. This topic – which we call 'security of inter-generational access to resources' or SIAR – emerged in the course of a series of multi-country field tests conducted by CIFOR on the topic of C&I for sustainable forest management (SFM). In our forest-level investigations into the human condition, which we considered important for SFM, we found several reasons for interest in SIAR. One reason was the ubiquity in tropical forests of confusion and conflict relating to resources, including ownership, access rights and sharing of forest benefits. Social benefits from forests are highly diverse, for example access to fuelwood is sometimes critical (Fig. 5.1). A second reason was the unanimous agreement of numerous interdisciplinary field teams that SIAR was an important component of human well-being. Because of this general agreement about the importance of SIAR, CIFOR conducted additional research focused specifically on this issue.

In this chapter, we focus on SIAR, and our attempts to assess it. Recognizing the dramatic difficulty of assessing these issues, our overall approach has included: (i) frequent iterations; (ii) regular field testing; and (iii) extensive collaboration. Each of these is discussed briefly in turn.

Fig. 5.1. Fuelwood is one of the globally significant outputs from forests, and its local availability can have a major influence on the lives of many people. Here a family collect eucalypt wood for domestic use in Zimbabwe. Indicators are needed to define the sustainability of rural fuelwood use, including effects on communities.

The iterative element of our research has included a cyclical oscillation from conceptual thinking and literature review, to field testing and review of results, then back to conceptual thinking and literature review, in an ongoing process. In our attempts to understand and assess SIAR,[1] we began with desk work (review of our experience and of social science methodological literature). In June of 1996, Colfer and Wadley conducted a pre-test of eight methods, three of which addressed SIAR (Colfer *et al.* 1997a).[2] These results were evaluated, additional literature review was conducted, and a new methods binder was developed in January 1997, consisting of 12 methods (six of which pertained to SIAR).[3] The 12 methods were field tested in several locations in Cameroon and East Kalimantan, then revised again in January 1998 (resulting in advanced drafts of *The BAG*, *The Grab Bag* and *The Scoring and Analysis Guide*, Colfer *et al.*, 1999a,b; Salim *et al.*, 1999). These methods were tested again, systematically in Brazil and in East Kalimantan, and on an *ad hoc* basis in Gabon and the USA. Final revisions took place late in 1998.

CIFOR-sponsored field-testing has occurred in at least 27 communities in Kalimantan and Cameroon. Additional field-testing has occurred in Trinidad, Gabon, the USA and Brazil.

The process of methods selection/creation and C&I development has involved numerous and diverse collaborators. Our formal collaborators during this phase of the project have included researchers from Cameroon, Indonesia, Senegal, the UK, and the USA.[4] We have also sought disciplinary variety (including anthropologists, ecologists, agroforesters, community development specialists, and agronomists), in order to assess the utility of particular methods for people with various kinds and levels of training. Although all primary researchers had masters degrees or PhDs, all teams included members with lower levels of training (elementary school and up).

2 Security of inter-generational access to resources

In this section, we briefly discuss some of the literature pertaining to C&I on issues related to SIAR. We then present our current 'best bets' on related principles, criteria and indicators. We discuss the methods we consider most useful, as well as a possible scoring method for assessment purposes. We close this section with an example from our research of the analyses we are doing now.

A wealth of literature has been produced on indicators pertaining to human well-being. Michalos (1997), who seeks links among environmental, economic and social indicators,[5] notes the disjointed nature of research communities looking at indicators. In line with this problem, we realize that there may be relevant materials of which we are unaware. However, many of the sets of indicators developed in the West are not immediately usable in tropical forest areas, because of the absence of the statistics on which the indicators rely. Such reliance is abundantly clear from the examples provided in Michalos' discussion. Bossel (1996) has recently produced a paper based on a potentially more usable systems approach to indicators, though his *examples* also rely on statistics and other information rarely available.

Given the diversity of approaches to indicator development, we examine here only bodies of literature that are specifically relevant to access to resources, though from a variety of points of view. We have found useful materials from anthropology, timber certification, international processes, conservation, monitoring and evaluation, and community-based forest management.

The number of anthropological studies documenting the existence and functioning of local mechanisms that regulate access to resources is legion (particularly weighty contributions include those edited by Kunstadter *et al.*, 1978; McCay and Acheson, 1987; Fortmann and Bruce, 1988; Poffenberger, 1990; Croll and Parkin, 1992; Redford and Padoch, 1992; Padoch and Peluso, 1996). Ostrom's excellent (1990) analysis of institutions for collective action represents a growing body of political science literature on local management, including access to resources. Rose (1994) has produced a series of essays providing useful insights into the meaning of property itself.

However, when we turn to the question of assessing security of access to resources in timber concession areas, the number of sources dwindles dramatically. Many important actors in certification (e.g. Rainforest Alliance, 1993; Initiative Tropenwald (ITW), 1994; Schotveld and Stortenbeker, 1994; Soil Association, 1994; Lembaga Ekolabel Indonesia (LEI), 1997; Forest Stewardship Council (FSC), 1998; Higman *et al.*, 1999) list issues pertaining to security of access in their requirements, but they remain silent on how to assess the issue.

The Southern Appalachian Assessment (1996) in the USA dealt with many of the issues we consider important. A common, earlier antagonism in the USA toward certification may have prompted the phrasing of concerns as broad questions rather than as C&I. With regard to SIAR, they identified important issues relating to employment, income and the relationship of these to timber production and other forest-related industries; population; ownership and uses of land; and values pertaining to land and its uses.

The international processes vary in their treatment of SIAR. The Montreal Process (1997) is quite explicit about customary and traditional land rights of indigenous people, as well as employment and other community needs. The African Timber Organization's set of C&I (still under revision) is, similarly, quite specific on recognition of traditional rights and sharing of benefits (African Timber Organization, 1998). The International Tropical Timber Organization (ITTO)'s earlier set scarcely mentioned this issue (1992), but more recent versions are much more complete (1997). The Helsinki Process (1993) was explicitly formulated for the needs of European forests, and does not address this issue in the direct way we have deemed important. It does, however, refer to changes in the rate of employment in forestry, notably in rural areas. The Ministry of Agriculture and Forestry (1996) in Finland has made some modifications to the Helsinki Process, which strengthen local access to resources somewhat (e.g. 'Secures clarity of rights related to forests (e.g. ownership . . .)', 'Secures special rights (e.g. reindeer husbandry) of the Saami people and local people'). Tarapoto's FMU level C&I (1998) do not discuss rights to land, but specify quality of life, impact of the economic use of the forest on the availability of forest resources of importance to local population, and amount of direct and indirect employment as important indicators of 'local socio-economic benefits'.

Our own views on SIAR were, of course, affected by the history of our research activities. Using a comparative study (Hahn-Schilling *et al.*, 1994) of existing sets of C&I, as a basis for selection, CIFOR tested the C&I sets put together by LEI, Rainforest Alliance, the Soil Association, ITW and DDB (Schotveld and Stortenbeker, 1994), between 1994 and 1996. Since then, we have continued to scan the literature for new ideas and approaches. Perez (1996) provides a nice overview of work done on C&I for certification, with a strong emphasis on SIAR-related issues. Copus and Crabtree (1996) suggest a matrix for indicators of socio-economic sustainability, which includes a number of economic issues that may be more crucial in a 'developed' country context (rural Scotland) than in the Third World contexts where we have focused. Indeed, there is a wealth of economic indicators and models (see Becker, 1997, or Ruitenbeek, 1998, for recent overviews of these) that can be linked to SIAR issues, though they have not been a focus until quite recently in our work.

In the conservation realm, Borrini-Feyerabend, with Dianne Buchan (1997) published a two-volume work aimed at improving our ability to achieve 'social sustainability in conservation'. Many of the SIAR issues we have addressed in the timber management context correspond with issues in these volumes, though from somewhat different perspectives. Their work has key questions, such as 'How do the natural resources of the conservation initiative contribute to the livelihood of local people?' or 'Does the conservation initiative affect access to land or resources and the control over them for one or more stakeholders?' They also offer indicators, with 'warning flags', pertaining to many of the issues we have also found important.

The International Union for the Conservation of Nature (IUCN) has been involved in a number of efforts to improve monitoring and evaluation relating to human well-being. Poffenberger (1996) has edited a series of case studies from India, Nepal, Canada, Panama and Ghana, and concludes, among other things, that access to forest resources is important in stabilizing resources, by enhancing community initiatives to protect forests against degradation. Jackson (1997) reports some of the tools and methods used in Southern Africa to evaluate collaborative management of natural resources, though the emphasis is more on participation issues than on access to resources, *per se*. Prescott-Allen's (1995) concept of a 'barometer of sustainability' is particularly appealing, though its only direct, SIAR-related issue is 'wealth and livelihood'. It includes the idea that 'human wellbeing is dependent on the wellbeing of the ecosystem' (IUCN, 1995a). In this system, indicators are considered context-specific and are selected by the users (IUCN, 1995b).

Much of the growing literature on community-based management is pertinent. Stevens (1997) uses C&I to assess the sustainability of a Turkish forest village ecosystem. From the SIAR perspective, he looks at household finances (income sources and amounts, indebtedness). He also recognizes a link between 'the state of the natural resource base, and the . . . social and financial indicators that depend upon them' (p. 30). The work undertaken by the Asia Forestry Network has included methods for assessing access to resources (e.g. Poffenberger *et al.*, 1992a,b), and their case studies address SIAR issues (e.g. Poffenberger and McGean, 1993a,b; Poffenberger, 1998a,b). Similarly, the Rural Development Forestry Study Guides provide useful methodological insights, a wealth of case material and 'lessons learned' (Carter, 1996; Hobley, 1996). A recent review by the Ford Foundation (1998) also stresses the importance of elements of SIAR (phrased as 'ascertaining actual benefits' and 'promoting the equitable distribution of benefits'). Increasingly numerous people and groups are looking at community-based management (e.g. at the East West Center in Honolulu; Fox *et al.*, 1995); RECOFT (Regional Community Forestry Training Center in Bangkok, Thailand); members of the International Association for the Study of Common Property (Lynch and Talbott, 1995; Poffenberger 1998a,b), and most of this work seems to include a focus on SIAR as an important issue.

Burford de Oliveira's C&I testing in four community-managed forests (Indonesia (two), Cameroon, and Brazil) involves attention to SIAR. Although she stresses the differences in her outcomes from those discussed below (Bogor, Indonesia, 1998, personal communication), we see considerable overlap in draft reports. The process of clarifying the overlap, or lack thereof, between C&I for community-based forest management and for timber operations is underway (cf. Ritchie and Haggith, 1998).

2.1 Social C&I 'best bets'

Over the past 4 years, we have produced a set of social C&I that we think covers the main issues for assessing human well-being. The C&I related to SIAR, listed in Box 5.1, represent one of three main issues. A second issue that we tested systematically is 'Concerned stakeholders have an acknowledged right and means to co-manage forests equitably'. The third issue, which we have *not* tested in any systematic way during our project, is 'The health of forest actors, cultures and the forest is acceptable to all stakeholders'.

At this point we should stress our continuing iterative view on these issues. Although considerable effort by various parties has gone into

Box 5.1. Forest management maintains or enhances fair inter-generational access to resources and economic benefits.

1. Local management is effective in controlling maintenance of and access to the resources[6]
   1.1. Ownership and use rights to resources (inter- and intra-generational) are clear and respect pre-existing claims
   1.2. Rules and norms of resource use are monitored and enforced
   1.3. Means of conflict resolution function without violence
   1.4. Access to forest resources is perceived locally to be fair
   1.5. Local people feel secure about access to resources
2. Forest actors have a reasonable share in the economic benefits derived from forest use
   2.1. Mechanisms for sharing benefits are seen as fair by local communities
   2.2. Opportunities exist for local and forest-dependent people to receive employment and training from forest companies
   2.3. Wages and other benefits conform to national and/or International Labour Organisation (ILO) standards
   2.4. Damages are compensated in a fair manner
   2.5. The various forest products are used in an optimal and equitable way
3. People link their and their children's future with management of forest resources
   3.1. People invest in their surroundings (e.g. time, effort, money)
   3.2. Out-migration levels are low[7]
   3.3. People recognize the need to balance numbers of people with natural resource use
   3.4. Children are educated (formally and informally) about natural resource management
   3.5. Destruction of natural resources by local communities is rare
   3.6. People maintain spiritual or emotional links to the land

developing this set, we do not believe that it can be viewed statically or as 'the ultimate set'. Continued testing and use will, we believe, result in improvements; and the question of 'sustainability' (which the C&I are designed to measure) is an eminently malleable concept that we expect to evolve as well. The C&I process, in our view, is very much aiming at a 'moving target'.

As is immediately clear on perusal of these indicators, most remain difficult (and in some cases, impossible) to quantify. All team members accepted the utility and desirability of finding quantifiable indicators. However, the conclusion to date is that insisting on quantification in contexts where the accuracy and replicability of measurements cannot be assured is less wise than relying on qualitative assessment. Bossel (1996) also argues that quantification is not necessary in all cases.

2.2 Assessment methods

We have selected several methods for assessing SIAR, combined with a scoring system, that has been tested a final time in Brazil and Kalimantan. We have divided the methods into two types: those that we consider usable by assessors with bachelor's level training in biophysical sciences (*The BAG*, Colfer *et al.*, 1999a); and those that we have found useful but that require higher levels of exposure to social sciences (*The Grab Bag*, Colfer *et al.*, 1999b). Both sets make use of *The Scoring and Analysis Guide* (Salim *et al.*, 1999) and *The Resource Book* (McDougall *et al.*, 1999).

An ideal set of methods would include one method for each indicator, which together would provide a clear understanding of the criterion for which those indicators were selected. However, we have not been able to do this. Instead, each method we have selected tends to focus on one or two indicators, with a high potential to provide information on related criteria and indicators, *in passing*. Many of our methods function as mechanisms for facilitating communication between the assessors and the local people. Such communication is essential because of the great, global variety of human systems, and our conviction that specifying particular, 'correct' management systems, or particular ways of facilitating sustainability, would be inappropriate, as well as ineffective. Instead, our 'output' bias encourages us to try to determine whether management *functions* are being performed, by whatever local mechanisms may exist. One of the interesting, unintended consequences reported in this methodological testing has been to raise the awareness of the local people in the test sites about the conditions of their environment and the various factors contributing to the problems they see.

The SIAR-related methods in *The BAG* include: Histo-ecological Matrix; Participatory Mapping; and Historical Transects of the Landscape. In *The Grab Bag*, we have included the Iterative Continuum Method (ICM), Benefit Sharing among Stakeholders: Pebble Distribution Method 1; and Access to Resources by Generation: Pebble Distribution Method 2.

The overall scoring system is quite simple, though it requires the assessor's intelligence and judgment. The range is 0–10, with 0 being the least sustainable and 10 being the most sustainable. The scoring procedure involves listing the criteria and indicators on a spreadsheet. The methods proposed above provide one important source of information, as does the assessors' informal conversations, systematic observations and secondary data. The assessor then lists case material and field data collected on the particular indicator, on the spreadsheet, as collected, for subsequent perusal. Toward the end of the field stay, the assessor examines the spreadsheet, and makes an overall assessment, based on these various kinds of information, and gives a score for each indicator. Information gaps, or apparently conflicting data, can also be identified at this time. Scores from indicators can be averaged to provide scores for criteria, as can criteria for principles. Colfer tentatively assigned weights, with SIAR receiving

40% of the total ‘human well-being’ score. We anticipate further field-based feedback on both the scoring method and the weights we have assigned.

2.3 Examining the links between SFM and SIAR

During our Phase I research (1994–1996), we learned of the near unanimity with which researchers – both social and biophysical – considered SIAR to be crucial for SFM and for human well-being. We too thought it was important, but there were niggling doubts about the relationship. Although the majority of our team members felt, for instance, that security of land tenure was crucial for SFM, a few argued that SFM was impossible so long as people had complete freedom to do what they wanted on their own private property. Some economists also argued that the most economically rational approach is to sell off the trees from forested land, given the long time horizons and consequent high discount rates.

We decided that some research was warranted on the question of causal links between SFM and SIAR. Our strategy has been to test our methods, insofar as possible, in areas where collaborators had longer-term experience, so that we could compare the results from our assessment methods with their fuller understanding of the local context (for reliability). Then we hoped that reliable results could be plotted against the forest conditions in the areas we selected. We explicitly included a range of forest conditions in our test sites, from ‘forest-rich’ to ‘forest-poor’. We defined ‘forest-rich’ as areas where there were islands of people in a sea of forest; and ‘forest-poor’, the reverse: islands of forest in a sea of people.

Analysis of these data is a task that we have just begun, since results continue to come in. We provide here one of our early sample analyses, coming from a pebble distribution method on ‘sharing of benefits’. We have results from two ‘forest-rich’ sites: Bulungan Research Forest in northern East Kalimantan (Sardjono *et al.*, 1997) and east of the Dja Reserve in eastern Cameroon (Tchikangwa *et al.*, 1998). The ‘forest-poor’ sites include Long Segar, in central East Kalimantan (Sardjono *et al.*, 1997); and the area around Mbalmayo in central Cameroon (Tiani *et al.*, 1997).

In Bulungan (Fig. 5.2), the primary stakeholders,[8] among whom benefit distribution was estimated, included the government, merchants, a foreign project, logging companies, two swidden cultivator groups (*Lun Daye* and *Abay*), a hunter-gatherer group (*Punan*) and ‘other’ ethnic groups.

Near the Dja reserve in Cameroon (Fig. 5.3), primary stakeholders included the government (*l‘Etat*), local government (*Autorités Administratives*), the forest service, the conservation projects, logging companies, merchants (*bayam-sellam*), traditional chiefs, village development committees, forest workers, a local ethnic group (*Nzime*), an immigrant ethnic group (*Kako*) and pygmies (*Baka*).

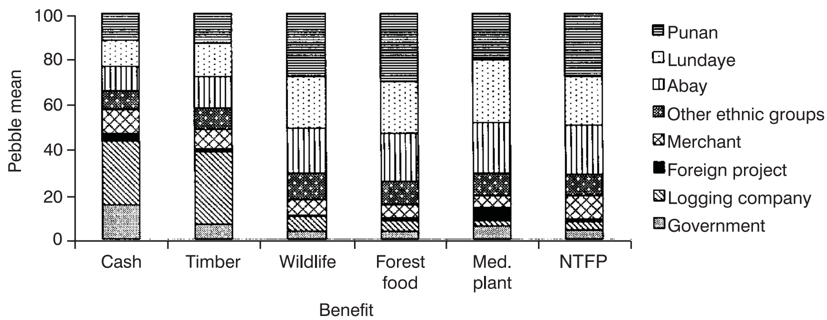

Fig. 5.2. Benefit sharing among stakeholders (all respondents), Bulungan, East Kalimantan.

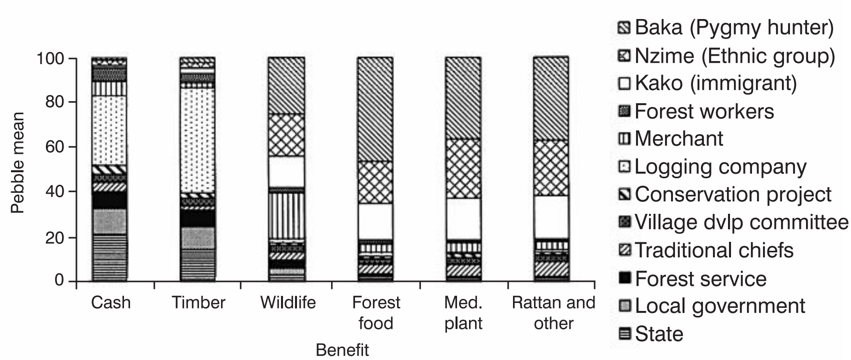

Fig. 5.3. Benefit sharing among stakeholders (all respondents), Dja Reserve, Cameroon.

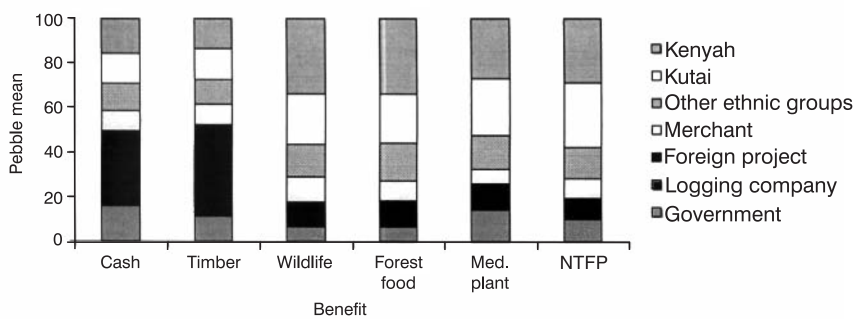

Fig. 5.4. Benefit sharing among stakeholders (all respondents), Long Segar, East Kalimantan.

Turning to the 'forest-poor' setting, in Long Segar, East Kalimantan (Fig. 5.4), stakeholders included the government, merchants, a foreign project, logging companies, two swidden cultivator groups (*Kenyah* and *Kutai*) and 'other' ethnic groups.

The Mbalmayo, Cameroon set (Fig. 5.5) included the government (*Administration Etatique*), logging companies, local merchants, sawmill workers, immigrants and the local population.

A striking, though not surprising, result is the dominance of governments and timber companies in access to cash and timber, as important forest benefits, on all sites. Similarly, in all sites the local people have dominant shares of the other forest products. If we compare the forest-rich sites (Dja Reserve and Bulungan) and the forest-poor sites (Mbalmayo and Long Segar), we find that the proportion of shares of wildlife, forest foods, medicinal plants and other non-timber forest products, perceived to be going to the government and the logging companies, rises somewhat in the forest-poor sites.

The people of the Dja Reserve reported very little of the area's cash and timber going to local people, when compared to local people's share in forest-poor Mbalmayo. Of course, the population density is also considerably less in Dja. This method does not help us understand any changes in overall availability of the respective forest benefits (though other methods we use do).

The local people in forest-rich Bulungan are seen to get a greater proportion of these valuable goods than do the people of forest-poor Long Segar. Although this requires further investigation, we suspect that this reflects an historical process observable in both countries. The area around the Dja Reserve, though rich in timber, may not yet have joined in full-fledged timber harvesting, whereas both Mbalmayo and Bulungan currently support an active timber industry. Long Segar is an area where timber has been extracted on a large scale since the mid-1970s, and the industry is turning to industrial timber plantations there at this time.

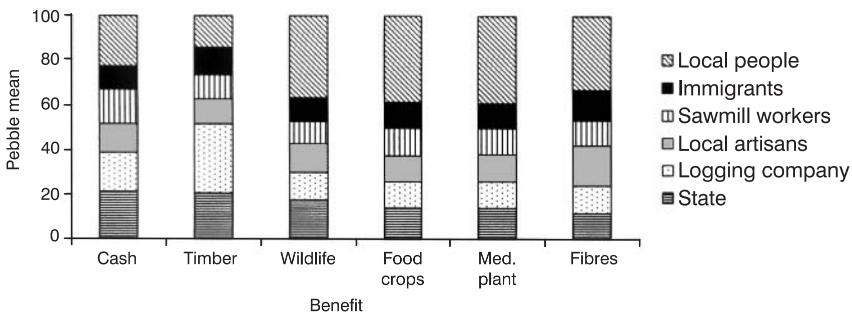

Fig. 5.5. Benefit sharing among stakeholders (all respondents), Mbalmayo, Cameroon.

We also use cluster analysis and biplots to show the degree to which there are dissimilarities between the forest-rich and forest-poor sites with respect to the benefit sharing pattern. Cross-country analyses involving sites with the same forest quality were made to see if there was any variation in the benefit-sharing pattern in different countries with the same forest quality.

Cluster analysis involves 'grouping' the stakeholders from sites with different forest quality within one country as well as 'grouping' the stakeholders from sites with the same forest quality across countries.[9] We analyse the within-country case first, to see if there are important dissimilarities in the allocation of rights to manage the forest in forests of different qualities.

Cameroonian site comparisons

Clustering stakeholders from two sites in Cameroon yields four clusters at final partition. The cut off point is 70% similarity. As can be seen in Fig. 5.6,

Cluster Centroids

Variable	Cluster 1	Cluster 2	Cluster 3	Cluster 4	Grand centroid
Cash	9.6467	31.6000	1.4000	22.5000	11.1222
Timber	9.1200	46.9000	2.3000	14.0000	11.1111
Wildlife	9.1267	1.4000	25.5000	36.2000	11.1111
Forest food	7.5800	1.4000	46.7000	38.3000	11.1167
Medicinal plants	8.2867	0.7000	36.4000	38.6000	11.1111
Other NTFP*	8.5867	1.2000	36.9000	33.2000	11.1167

* Includes ratten, fibres and other NTFP.

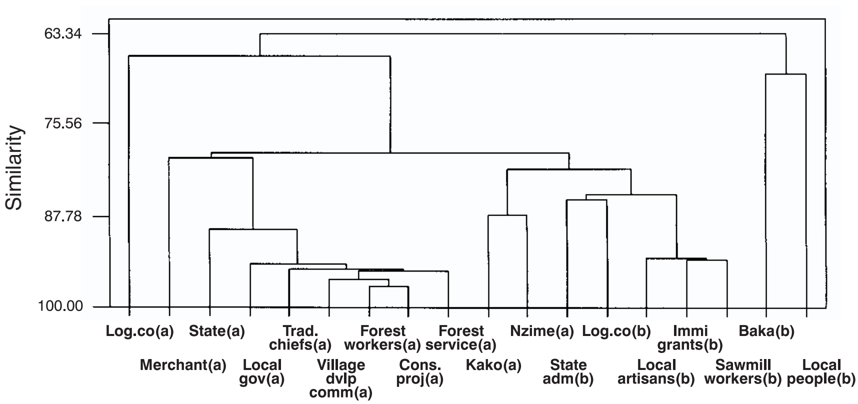

Fig. 5.6. Clustering stakeholders from two Cameroon sites, based on benefit sharing. (a) Stakeholder from forest-rich sites; (b) stakeholder from forest-poor sites.

cluster 1 contains 15 stakeholders, while the other stakeholders fall into the other three clusters, with one each. Cluster 2 contains the logging company from the Dja Reserve (a), cluster 3 contains Baka Pygmies (b) and cluster 4 contains local people from the Mblamayo site (b). Cluster centroids describe each cluster profile.

The interpretation of biplots is simple, stakeholders lying close to each other tend to have similar profiles. If the stakeholders and the 'variable arrows' go in the same direction, that stakeholder has been allocated a larger value (i.e. more rights) on that variable. Stakeholders with positions lying opposite to the 'arrow' direction show that they have been allocated lower values, or fewer rights, on that variable. The cosine of the angles between two vectors will represent the correlation between those variables (Jolliffe, 1986).

One initially notices that 'local people' (b) and the Baka (or pygmies, b) form clusters 3 and 4, to the right of Fig. 5.6. These represent the stakeholders that are perceived locally to be the most forest-dependent of those identified by the researchers in each dataset. Their perceived access to benefits is very clearly different from all other stakeholders in the two Cameroon contexts presented here. Within the left-hand cluster (2), the next lower cluster level includes the logging company in the forest-rich context, counterposed to the large group (cluster 1) that contains most stakeholders. That third level group is also divided in two, with one dominated by forest-poor stakeholders, such as more sedentary forest people, artisans and sawyers. The local people from the forest-rich area who are included in this category are the Nzime and the Kako. Both ethnic groups live in a community that was specifically selected as an atypical village (Sembe). The Nzime are the original inhabitants of Sembe, which is now occupied primarily by newcomers, with the numerically dominant Kako coming from the forest–savannah border to the north. These two ethnic groups represent the only stakeholders from a forest-rich setting that occur in this cluster. Contrasted at this same level is a wide range of forest-rich stakeholders, including forest workers, traditional chiefs, the state, merchants, etc.

The biplots show the perceived distribution of benefits dramatically. The logging company at the forest-rich site (Dja Reserve) is perceived to receive the dominant share of cash and timber, while the share of the logging company in the forest-poor site (Mbalmayo) is less strikingly dominant. Baka Pygmies are perceived to receive the dominant share of forest foods, medicinal plants and other non-timber forest products, and also to have a fairly good share of the wildlife. Local people from Mbalmayo (b) are perceived to receive a greater share of wildlife and also to have a good share in the other benefits, excluding timber. We may conclude that local people in the forest-poor area are perceived to have a larger share of *cash* than people at the forest-rich site. The biplot (Fig. 5.7) also shows the State at Mbalmayo to have a bigger share of everything than in the Dja Reserve. The logging company gets more benefits, especially cash and timber, from the forest-rich site.

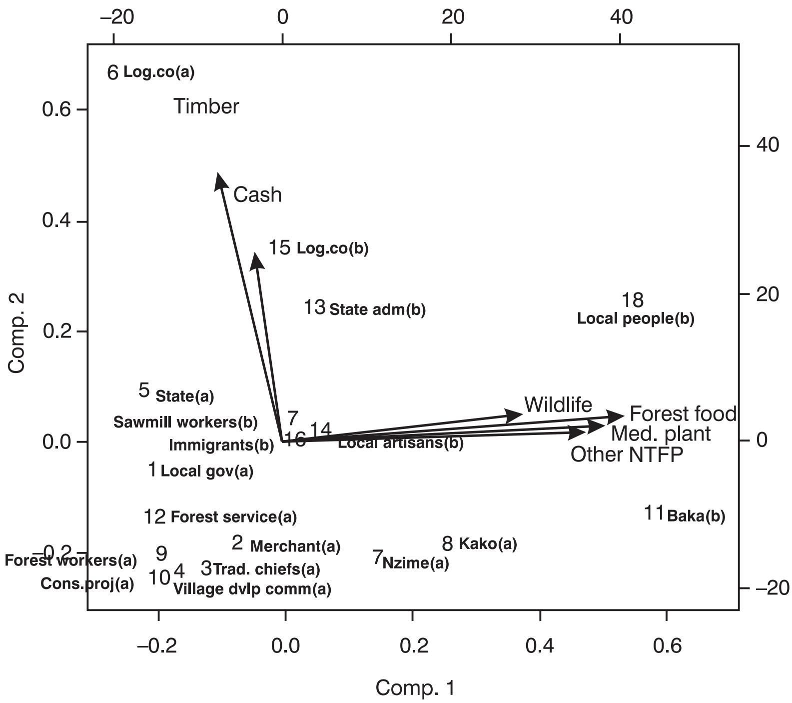

Fig. 5.7. Biplots showing benefit sharing among stakeholders at two Cameroon sites.

Indonesian site comparisons

With fewer stakeholders, clustering stakeholders from two sites in East Kalimantan yields two groups as the optimal size (Fig. 5.8). The second cluster, containing the logging company from each site, has a bigger share of cash and timber than the other stakeholders do. The first cluster contains the rest of the stakeholders, who, although seen as receiving a smaller share of cash and timber, are still perceived to have a bigger share of the other benefits (fruit/vegetables, medicinal plants, wildlife and NTFPS). The biplot (Fig. 5.9) shows stakeholders with similar roles, lying close to each other, reflecting the fact that the differences between the forest-rich and the forest-poor sites in Indonesia are not as big as those in Cameroon.

In Indonesia, the logging companies in both sites form cluster 2, on the right side of Fig. 5.8. Cluster 1 comprises the other stakeholders, further divided neatly into local people from both sites, on the one hand, and all other stakeholders (including ‘other groups’, or in-migrants) on the other. The reliability of these internal differentiations – within clusters – is lower than that between clusters of course.

Cluster Centroids

Variable	Cluster 1	Cluster 2	Grand centroid
Wildlife	14.3231	6.9500	13.3400
Cash	11.1385	27.6000	13.3333
Fruit/veg	14.4308	6.2500	13.3400
Medicine	14.7769	4.0500	13.3467
NTFP	14.7385	4.2000	13.3333
Timber	10.1385	34.1000	13.3333

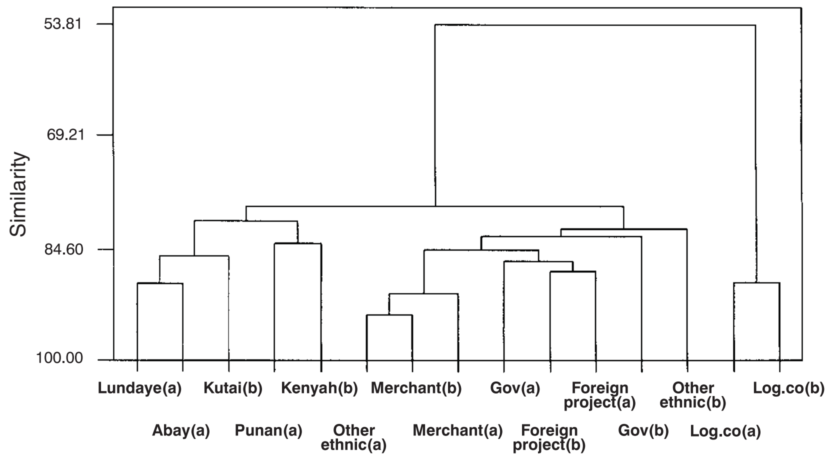

Fig. 5.8. Clustering stakeholders from two East Kalimantan sites based on benefit sharing.

The biplot shows the division of money and timber for the logging companies, contrasted to other forest products for local people very clearly, with considerably lesser benefits to other stakeholders.

FOREST-RICH COMPARISONS Looking at cross-country comparisons, the stakeholders from the two forest-rich sites yield three clusters at the final partition (Fig. 5.10). Cluster 1 contains 17 stakeholders, cluster 2, two, and cluster 3, one stakeholder. Members of cluster 2, on the left side of Fig. 5.10, are the logging company from each site. Those companies are perceived to have a large share of cash and timber but less of the other benefits. In this way, these companies' profile is very similar. Cluster 3, to the right in Fig. 5.10, contains Baka Pygmies, an ethnic group from the Dja Reserve perceived to receive a dominant share of the other benefits, excluding timber and cash. This group uses the forest in quite different ways from the other more sedentary groups. The fact that the Indonesian Punan, another traditional hunting and gathering group, are not perceived to be in the same cluster is interesting, and may be explained by their recent resettlement in Paking, where they have been urged to adopt a more sedentary lifestyle. The other 17 stakeholders, in the

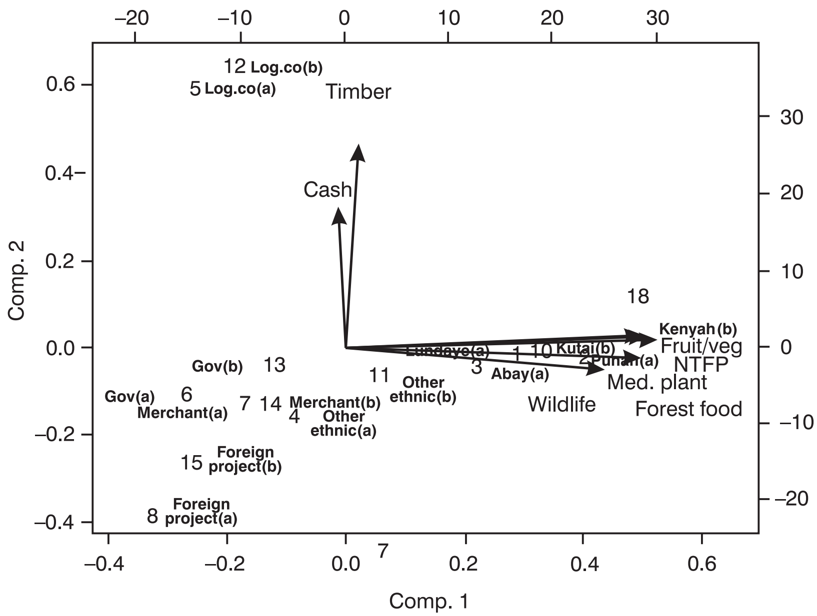

Fig. 5.9. Biplot showing benefit sharing among stakeholders at two East Kalimantan sites.

centre of Fig. 5.10, are seen to have a smaller share of these benefits. As before, we have an internal differentiation between local people (centre right) and other stakeholders (centre left), like the government, merchants, forest workers, etc. This internal differentiation was also not strong enough to create additional clusters.

From the biplot (Fig. 5.11), we can see that no local ethnic group in the Dja Reserve is perceived to have a good share of timber and cash. This is clear from the position of the Baka, the Nzime and the Kako 'behind', or in the opposite direction of the arrows for timber and cash. The local ethnic groups from Bulungan (Punan, Lundaye and Abay) are seen to have a better share of cash and timber. Also we can see that although the logging company from both sites falls into one cluster, the Cameroonian company is seen as receiving a more dominant share of cash and timber than the Bulungan company.

FOREST-POOR COMPARISONS The forest-poor sites give two clusters as the optimal size, with the cut-off point at 70% (Fig. 5.12). Cluster 1, to the left of Fig. 5.12, contains ten stakeholders and cluster 2, to the right, contains three. Members of cluster 2 are local people/ethnic groups in those areas, with 'local people' (C, coming from Cameroon), Kenyah and Kutai (I, coming from Indonesia). These local ethnic groups are strongly characterized by their large perceived share of wildlife, forest foods, medicinal plants and other NTFPS.

Cluster Centroids

Variable	Cluster 1	Cluster 2	Cluster 3	Grand centroid
Cash	8.1588	29.9500	1.4000	10.0000
Timber	6.9765	39.5000	2.3000	9.9950
Wildlife	9.8118	3.8500	25.5000	10.0000
Forest food**	8.6647	3.0500	46.7000	10.0050
Medicinal plant	9.4529	1.5000	36.4000	10.0050
Other NTFP *	9.3118	2.4500	36.9000	10.0050

* We combine the terms," NTFP " from Indonesia and "Fibres, Rattan" from Cameroon, in this category.

** We combine the terms, "Fruit/vegetables" from Indonesia and "Forest food" from Cameroon, in this category.

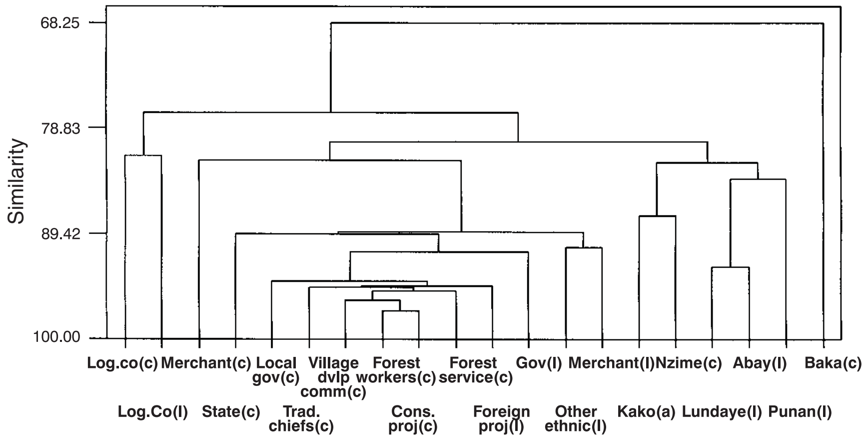

Fig. 5.10. Clustering stakeholders from two forest-rich sites based on benefit sharing.

None of these ethnic groups is as strongly forest-dependent as the Baka or hunter– gatherer Punan. The inclusion of the timber companies in the same category with other stakeholders like the state, workers and merchants represents a potentially interesting difference from the forest-rich context.

From the biplot (Fig. 5.13), we can see that logging companies from both sites are perceived to receive the dominant share of cash and timber, with the Indonesian timber company perceived to be getting a very dominant share of timber (although less than in forest-rich sites). The government at Mbalmayo (*Etat Administrative*) is perceived to have a considerably bigger share of cash and timber than other stakeholders at Long Segar, including the government.

We expect to do more in-depth comparative analyses of these data, looking for instance at the perceptions of women *vis-à-vis* men, of the different stakeholders, and at the congruence, or lack thereof, among different stakeholders' perceptions. We will also analyse the results from our other methods tests, to see what patterns emerge in the forest-rich as opposed to the forest-poor sites, that can help us to improve our understanding of the links between sustainable forest management and SIAR.

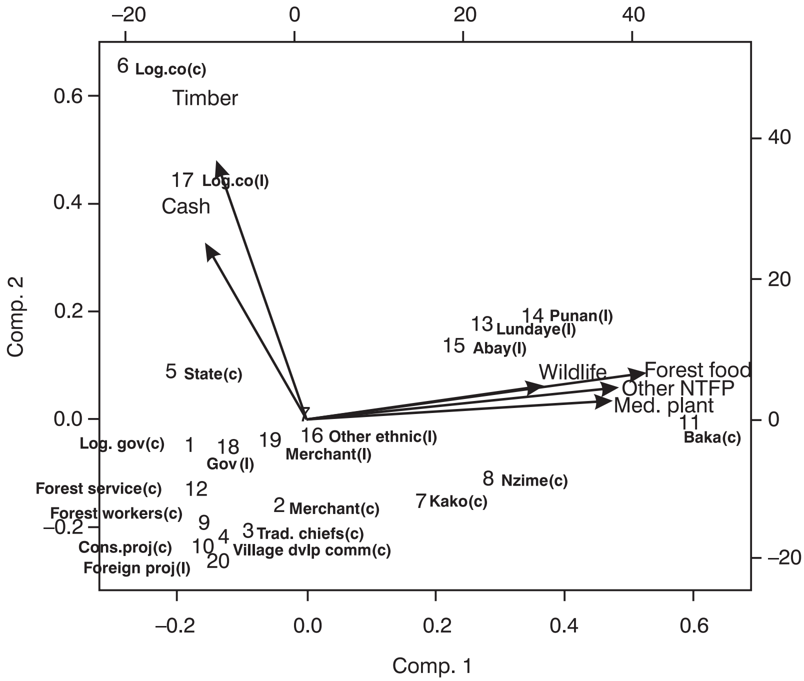

Fig. 5.11. Biplot showing benefit sharing among stakeholders at two forest-rich sites.

3 Future research needs and plans

We see four major issues that still need attention in our own work; and we hope that others will join us in addressing them.

First, we want to complete our analyses on causal links between SFM and SIAR. We have a vast amount of data from our methodological tests, that needs to be more thoughtfully and thoroughly meshed with our collaborators' long-term, qualitative understanding of local conditions. We also need to obtain fuller information on the environmental conditions in the sites. Although we have been able to plot our locations on a rough continuum of environmental quality, we would like to have the kind of careful, remotely sensed, time series data that we have for our West Kalimantan sites (1972, 1990 and 1994; Dennis *et al.*, 1998). The relationships between SFM and SIAR are important if we are to be able to refine the C&I that we have developed.

Second, we want to fill in the gaps in the existing C&I pertaining to the third important issue we identified relevant to human well-being: 'The health of forest actors and cultures is acceptable to all stakeholders'. As the related C&I now stand, they address important issues, but they contain overlaps with

Cluster Centroids

Variable	Cluster 1	Cluster 2	Grand centroid
Cash	14.8300	17.3000	15.4000
Timber	15.8600	13.8333	15.3923
Wildlife	10.6900	31.0667	15.3923
Forest food	10.5700	31.4667	15.3923
Medicinal plants	10.8500	30.5333	15.3923
NTFP	10.8800	30.4000	15.3846

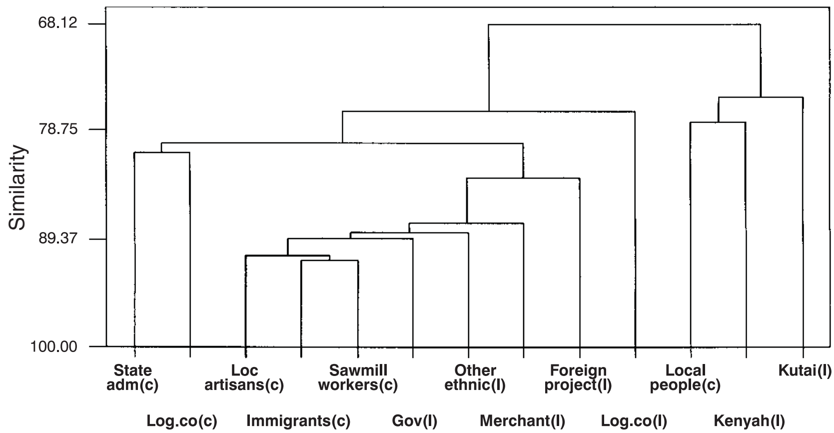

Fig. 5.12. Clustering stakeholders from two forest-poor sites based on benefit sharing.

some of our other C&I, and they have not been field-tested at all. We think further thought, literature review and field-testing are important for fine-tuning these C&I for wider use.

Third, we have found gaining access to certain segments of forest communities to be difficult, despite our best efforts to get representative samples. As mentioned above, good communication between the assessors and the local people is absolutely essential to a good assessment. In one locale, a team had trouble gaining access to anyone but the aristocracy (Burford de Oliveira, 1997). In all of our sites we have had more trouble gaining access to women's points of view than men's (Colfer *et al.*, 1997b). In all of our test sites, women have also had important, forest-related roles and responsibilities, suggesting that their input is crucial for evaluating both SFM and human well-being.

Finally, and most ambitiously, we think that the next step for CIFOR should be a concerted attempt to implement C&I-based monitoring, in pursuit of actual, sustainable forest management. As Prabhu *et al.* (1998) have discussed, we expect to identify several field sites to monitor using our C&I template (or 'best bets' C&I). We will work closely with local stakeholders (whether community managers, logging companies, conservation area managers, etc.), facilitating their adaptation of the C&I to develop a scientifically sound,

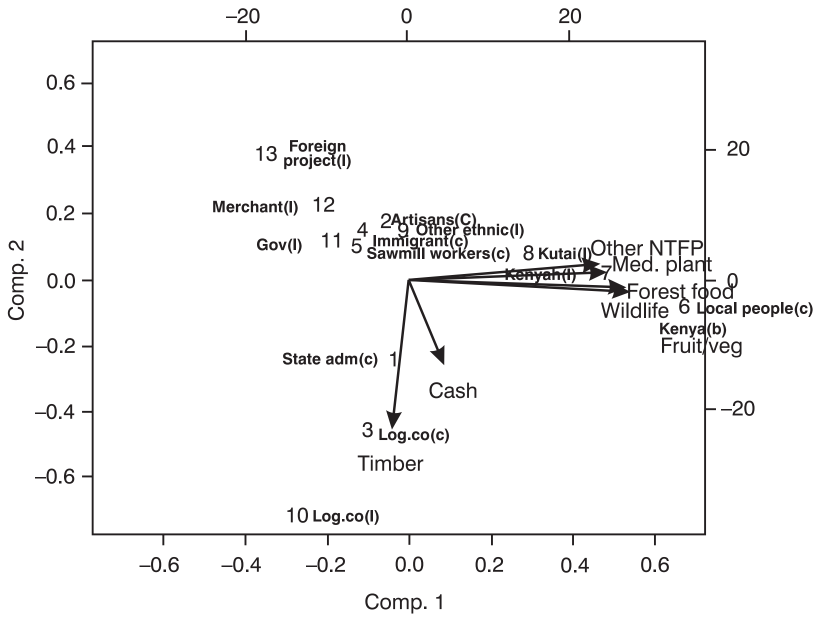

Fig. 5.13. Biplot showing benefit sharing among stakeholders at two forest-poor sites.

cost-effective and simple monitoring arrangement. In the biophysical sphere the system will allow the users to register, analyse and respond to impacts, for instance, on regeneration or soil. In the social realm, we anticipate the need to improve integration of relevant stakeholder subgroups, especially marginalized groups, into management decision making (something also called for by Shindler *et al.* (1996) and Stankey and Clark (1998).

Our research interests are threefold: understanding the adaptive learning and decision-making processes that are at the core of sustainable forest management, improving our understanding of stakeholder integration into decision making and carrying out improvements to C&I in an action research context. Some of our tools will include conflict resolution strategies (Resolve, 1994; Chandrasekharan, 1996; Ramirez, 1998); 'Future Scenarios' (Buck and Wollenberg, 1997), a tool for bringing together diverse interests in decision making; MAS (Multi-Actor Systems), a computer tool for enhancing interaction among stakeholders (Cossalter, 1997; Guizol, Bogor, Indonesia, 1997, personal communication); and stakeholder agreements about uses (e.g. Mayers and Kotey, 1996; Nguinguiri, 1999).

We will also identify and build on links between local conditions and policies that affect SFM and human well-being, using techniques we identify in planned literature reviews and case studies around the world, and of course continue our work on C&I related to the health of people, forests and cultures.

Notes

1 The process of concluding that SIAR was important was also an iterative process (beginning with our conceptualization of the problem (Colfer *et al.*, 1995), and several field tests with subsequent analysis of results (Prabhu *et al.*, 1996)).
2 These included a History Form: Iterative Continuum Method (ICM): and Participatory Mapping.
3 These included Historical Trends Analysis; Historical Transects of Landscape; ICM; Participatory Mapping; Benefit Sharing among Stakeholders: Pebble Distribution Method I; Access to Resources by Generation: Pebble Distribution Method 2.
4 Bertin Tchikangwa, Anne Marie Tiani, Mustofa Agung Sardjono, Chimère Diaw, Mary Ann Brocklesby and Reed L. Wadley, respectively.
5 The CIFOR work has included ecologists, forest managers and social scientists in many field tests; and an effort has consistently been made to put the results together into an integrated fashion (most recently, CIMAT, or C&I Monitoring and Adaptation Tool; Prabhu *et al.*, 1998).
6 This criterion is obviously very closely connected with criteria addressed from ecological and formal 'forest management' perspectives.
7 Indicators 3.2 and 3.1.2 contain a potential contradiction. Low levels of out-migration (3.2) indicate that people link their and their children's future to maintaining the forest; yet recognizing the need to balance numbers of people with natural resource use (3.1.2) may lead them to favour out-migration. Indicator 3.1.2 is in the ecological section of the set and not reproduced here. This contradiction would likely occur when conditions are deteriorating.
8 We are defining stakeholders as individuals/groups having an interest in the forest. Part of our testing process included identification of stakeholders (discussed in Sardjono *et al.*, 1997; Tiani *et al.*, 1997; and Tchikangwa *et al.*, 1998; see also Colfer, 1995).
9 A hierarchical clustering with single linkage (nearest neighbouring) method was used since there are some outlier values in the data. Lance and Williams (1967) pointed out that in the presence of outliers, the single linkage method was virtually unaffected, while Ward's method and group linkage methods performed poorly. For determining the number of groups in the final partition we used Mojena (1977, revised by Milligan and Cooper, 1985). Details of the formula can be seen in Everitt (1993).

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Representing the Future: A Framework for Evaluating the Utility of Indicators in the Search for Sustainable Forest Management

6

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The debate about sustainability of forest land management, coalesced by the Brundtland Commission report in 1987 but triggered by growing concerns about the impact of human activity on the environment, serves the important functions of requiring society to think about the future, consider its relations with nature and explicitly deliberate the consequences and implications of the resource development choices it makes. Fundamentally, sustainability is about equity and inter-generational justice, and thus there is a need to periodically assess progress toward these goals by use of specialized variables termed indicators. The indicators that are chosen reflect social decisions about what it is that should be sustained – a political choice but one informed by science. The pursuit of suitable, measurable indicators, however, is influenced by how citizens, managers and scientists address three fundamental issues: (i) the character of sustainability; (ii) institutional arrangements upon which the pursuit of sustainability is based; and (iii) tests for evaluating the usefulness, measurability and validity of variables proposed for indicators.

1 Introduction

The growing international interest in the issue of sustainability derives from two basic beliefs. First, there is spreading concern that current levels of

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resource utilization, consumption and modification, if continued unabated into the future, will lead to irreparable harm to both bio-physical and socio-economic systems; at worst, failure to modify our patterns of consumption could lead to catastrophic collapse of these systems. Second, and clearly related to the first, there is a strong moral belief that present-day patterns of resource utilization unfairly and inequitably limit the options and choices for future generations. Accordingly, a host of global, national and regional protocols and agreements, along with extensive discussion in the literature, have embraced the concept of sustainability as an essential, integral component of resource management. For example, sustainable forest management (SFM) symbolizes efforts to identify and deal explicitly with the trade-offs between the needs of current and future generations. This is a particularly relevant issue for forest managers, given the inherent long time-frames involved in forest rotation, client preferences and changing public values.

The search for sustainability, however, has proven frustrating and elusive. For example, although the normative belief of preserving future options is generally acknowledged as both important and legitimate, operationalizing such a perspective is extraordinarily difficult, not only because the institutional framework to do so is largely absent, but also because a consensus on what particular mix of goods and services from forests future generations might desire is largely non-existent.[1] Despite the generally accepted ethical belief that we have a responsibility to future generations, our inability to understand what it is we should do today to accommodate the unknown demands of tomorrow makes the pursuit of sustainability problematic.

However, we would hasten to add that concern with sustainability and SFM are more than hollow phrases or empty rhetoric. Sustainability embraces what has been called a guiding fiction (Shumay, 1991), i.e. precepts that cannot be proven or measured, but which serve to create a sense of community, connection and purpose, that serve to trigger necessary political and social discourse. However, an irony of guiding fictions is that the pursuit of specificity, clarity and measurement can operate to counter the principles underpinning the concept. Too often, agreement at the conceptual level and in shared meanings collapse as self-interest outweighs the collective good. Thus, debate about sustainability serves the important function of requiring society to think about the nature of the future, about its relation to nature, and to consider explicitly the consequences and implications of the alternative choices which confront it. This is a messy business and, as scientists and technical experts, we must take care to recognize the inevitable social choice issue which sustainability constitutes, while offering the kind of specialized expertise and knowledge such choices must accommodate.

Sustainability is fundamentally concerned with the goals of equity and inter-generational justice. The pursuit of SFM thus involves how forests can be managed to achieve these goals. Understanding how well our pursuit is going requires periodic assessment of the condition of forests through the use of specialized variables termed *indicators*. Indicators perform important functions in forest management by helping us to understand the degree of progress toward achieving a goal. The choice of these indicators – and guiding criteria – is influenced by a variety of factors including the specific determination of what is to be sustained. Since the choice of goals is a political one, the selection of appropriate criteria and indicators (C&I) of SFM must accommodate both the technical expertise of forestry specialists while also being informed by fundamental beliefs in equity and justice. The need to accommodate both dimensions is central to our argument for the need for a new paradigm of management to achieve sustainability.

In light of these concerns, we explore three questions critical to the selection of appropriate and suitable indicators of SFM. First, we examine the character of sustainability and SFM. The Brundtland Commission (World Commission on Environment and Development (WCED), 1987) did much to encourage discussion of global environmental issues and the relation of these issues to the distribution of wealth, pollution and access to political opportunity. However, its report and especially the discussion of sustainable development has contributed to a sense that existing paradigms of natural resource management and economic development need only to be adjusted at the margin in order to promote policies and actions that are ecologically viable, socially acceptable and economically feasible; in short, that allow us to 'meet the needs of the present while ensuring future generations options to meet their needs' (WCED, 1987, p. 40). In contrast, we argue that the conception of sustainability and SFM as being dependent upon simply refined technical and economic processes is not only incorrect, but such a fundamental miscasting of the problem as to make effective implementation impossible. As Socolow (1976) and Wondolleck (1988) have noted, technical analyses fail when asked to resolve questions of value judgement, and while science plays a critical role in the selection of indicators, it can only inform, not determine, their selection.

Second, we propose the types of institutional changes needed for a more effective pursuit of SFM and the selection of relevant, appropriate and suitable indicators. While our first argument could logically lead to the conclusion that the current partnership of democracy and free-market economics is not viable to achieve SFM, we are nevertheless confronted with the reality of not only the enormous success of these two institutions but the lack of acceptable alternatives. Given this situation, we suggest the types of institutional changes needed to better achieve SFM. These newer institutional arrangements would flow directly from agreements on what forest management should sustain. Because sustainability is as much a moral question as a scientific one, progress can be made only if the variety of interests at stake are routinely and legitimately involved in the selection of C&I.

The third section proposes a set of tests for evaluating the usefulness of C&I of SFM for the individual forest unit level. SFM has several meanings and the selection of C&I must reflect those meanings. Our view of forest management is broad, including not only forests managed for resource commodities but areas managed for a host of other values, such as biodiversity and amenity uses. We submit that C&I cannot be selected without understanding the question of what should be sustained, or the institutional context within which forest management occurs.

2 SFM: a wicked problem, a messy situation

The pursuit of SFM is founded on agreement on what should be sustained. Agreements are intrinsically a matter of political and social preferences, and while influenced by technical information and concepts, are a function of a variety of processes including the distribution of power, the presence of structural distortions in the availability of information, and the degree to which basic values are shared among various stakeholder groups. Given the contentious nature of forest management globally, definitions of SFM may not be widely shared, nor may they be apparent. The resolution of the *wicked* character of SFM thus leads to important roles for stakeholders.

As generally represented, sustainability requires: (i) achieving a variety of social, economic and environmental goals; (ii) more or less simultaneously while; (iii) providing options for the future; yet (iv) meeting the needs of the present. However, in reality, operating across these variable conceptions of outputs, over differing spatial and temporal scales, in the context of a pluralism of public tastes and preferences, makes any ready definition of what constitutes SFM problematic. Moreover, the goals at which SFM might be directed vary significantly in terms of their explicitness, measurability and compatibility.

In developing goals for SFM, one must incorporate the decision-making context, which can be described by a number of variables. There may be a tendency to assume that this context is relatively simple. Traditional synoptic or social-reform planning models (Friedmann, 1987) function best where problems are well defined (with general agreement on the definition among publics), study protocols are standardized (scientific method), experimental conditions can be manipulated and controls are in place. Assumptions about the validity of these characteristics are often hidden or implicit at best, yet they strongly influence the processes used to plan and make decisions.

In these simplified settings (which may be termed tame problems), policy makers have perfect information, infinite time and clear choices (Forester, 1989). These formal, theoretical conditions, however, are seldom encountered in the turbulent, ambiguous and contentious setting in which natural resource management occurs. Conflicts are often more about values – often vaguely specified – than specific actions to attain agreed goals.

The pursuit of SFM typically takes place in environments where competing interests strategically contend for attention, and multiple, often incompatible, values are desired; where the impacts of our actions (or inactions) are poorly understood and/or distributed inequitably. Both time and information are scarce resources; and the inequitable distribution of either among competing constituents can convey power to one interest over another.

Indeed, a radical critique of sustainability would go further, arguing that SFM cannot occur in settings where political and economic relations are structurally distorted, and where the problem (achieving sustainability) is ideological, not scientific. Dovers and Handmer (1993) maintain that sustainability is characterized by deep-seated contradictions ‘between perhaps irreconcilable goals and directions’, and the concept of sustainable development has been criticized as a mainstream capitalist response to environmental problems seeking simply a ‘more enlightened approach’ to resource management problems (Jacob, 1994). Robinson (1993) argues that mainstream notions of sustainable development never critically examine the process of economic development.

It is against this backdrop that discussions of SFM must be framed. Forestry (in its narrowest meaning) deals with managing the natural capital or biological assets of a forest. However, protecting natural capital (an environmental goal that provides options for the future) requires that we both understand and agree on what natural capital should be safeguarded. For example, Callicott and Mumford (1997) argue that our concern with the management of natural capital in ‘humanly occupied and exploited’ settings is primarily focused on issues of ecosystem health (which briefly stated, may be said to occur when ecosystem processes occur, but desirable species may have replaced naturally occurring ones), while in biological reserves, attention shifts to questions related to ecological integrity (maintenance of historic species composition and structure). While these are admittedly imprecise concepts, there is little question that they are different and that they lead to different emphases in terms of the indicators one might use to monitor and evaluate forest management. Yet, we can’t help but wonder about the extent to which there would be agreement over the relative merit of these alternative emphases as goals for SFM among professional foresters, scientists and different public interests (let alone agree about the appropriate management strategies for either goal!).

The above two goals offer an example of the need to identify what it is we wish to sustain. We submit that there are a number of choices, as Gale and Cordray (1994) have argued. They outline nine possibilities, ranging from undisturbed ecosystems to yields of highly valued products. The choice of what to sustain in forest management is a fundamental one, and one that is ascertained through political, not scientific, processes. In order for these political decisions to be socially acceptable, they must incorporate the interests and values of various stakeholders. Thus, while selecting indicators may seem to be within the realm of science, choices are conditioned on informed political deliberation about what to sustain.

There are two important conditions that contextualize the decision with respect to indicators. First, scientists must agree on cause–effect relationships. This is a question of validity. Understanding causes and effects is a prerequisite to selecting indicators that are useful in guiding management actions (causes) and which will provide reliable measures of progress toward achieving goals.

However, the levels of scientific agreement on causation, particularly at the temporal and spatial scales relevant to the concept of sustainability, are arguable. For example, in the Pacific Northwest of the USA, the underlying causes of the decline of the salmon fishery are still hotly disputed among scientists as well as among various interest groups. While an adaptive management model has been proposed as a means of operating in the face of high levels of uncertainty (e.g. Holling, 1978; Lee, 1993), there are few reports of successful large (landscape level) experiments (Gunderson *et al.*, 1995). The lack of adequate experimental evidence means uncertainty remains high with respect to sustainable forestry, while the potential for surprise is a virtual certainty.

The second condition requires that there is political agreement (consensus) on what should be sustained. This agreement is needed primarily to understand where we want to go, and to marshall the funding, personnel and legal resources necessary to implement actions. However, the contentiousness that characterizes forest management throughout the world today suggests that this necessary political consensus is rare. Because indicators ideally should reflect management outputs, their selection presupposes agreement on what will be sustained, yet in light of the contentious nature of the forest debate in most countries, such an assumption seems unwarranted.

The lack of scientific agreement and public consensus about the goals of SFM force us to conclude that we face a *wicked* problem (Allen and Gould, 1986), one which can only be resolved through social processes involving venues that foster dialogue, encourage deliberation and enhance learning. Such processes require equal participation by scientists, managers and public interests but, as we discuss further shortly, are most notable by their absence on the institutional scene.

Mistaking a wicked problem for a *tame* one (i.e. a problem subject to resolution through conventional rational analysis) leads to two important consequences. First, the problem remains resistant to our efforts to resolve it because our attention is focused on symptoms rather than underlying causal factors. Caldwell (1990) has examined this issue, concluding that we have a tendency to frame many of our environmental management problems as simply the result of operational shortcomings, e.g. inadequate environmental assessment procedures, insufficient date bases, etc. To the contrary, he argues that many of these problems derive from fundamental flaws in the underlying techno-economic systems; resolving such systemic impairment requires basic changes in the technical and behavioural systems and institutions that govern society. As we consider the issue of SFM, it might be that we need think of it as systemic, rather than operational in character.[2] It is systemic because our inability to achieve sustainability flows from fundamental social, economic and political processes which combine to produce demands on ecological processes and functions that exceed the environment's restorative capacity over the long run.[3]

The second consequence is that the failure to identify the fundamental underlying nature of the sustainability challenge can result in significant opportunity costs in terms of the mis-allocation of financial, personnel and time resources. Treating sustainability as a computational problem, subject to resolution through conventional rational analyses (i.e. treating sustainability as a tame rather than wicked problem) could lead to investment of significant resources in data collection or development of new administrative procedures, when other, more radical changes are required. Failure to define adequately the nature of the question can lead to work at the wrong spatial and/or temporal scale, or through institutions poorly equipped to deal with the complexities that characterize sustainability. Moreover, diminished capacity results in more than spent money and used-up people – it challenges the very credibility of the formal institutions created to manage our forests.

3 Institutional characteristics necessary for SFM

Can sustainable forest management, however problematic the term, be achieved within existing institutional frameworks? This is an important question because to a large degree the concern over sustainability has arisen largely out of the perceived deficiencies of current law and agency policy. It might prove unrealistic to expect the system that caused the problem to now be given responsibility for resolving it. We turn to three brief examples to illustrate our concerns as to the capacity of current institutional structures and processes to implement SFM.

First, forestry agencies commonly use economics and discount rates to value the flow of revenue projected to occur at different points in the future. Through the use of interest rates, such revenues are discounted in value to what the current generation would prefer. The use of discount rates is based on the central assumption that people prefer revenue now rather than later. However, positive discount rates create a systematic bias that potentially understates the value future generations might place on various goods and services in comparison to the preferences of the current generation. The discount rate thus favours the needs of the present over protecting options for the future, a situation inconsistent with the Brundtland Commission's widely accepted definition of sustainable development.

Second, our experience in the USA suggests that much of the founding legislation for forest and land management is narrowly written and often antithetical to the goal of sustainability. For example, federally administered lands of the Forest Service and Bureau of Land Management frequently are subject to the 1872 mining law, which gives claimants absolute (and inexpensive) rights to sub-surface minerals. The exercise of these rights can devastate surface conditions (and a host of other values), leaving gaping holes, acid waste water and heavy metals for future generations to restore – hardly a natural heritage offering equal or better options. Such laws (referred to by Wilkinson (1992) as the 'lords of yesterday'), while socially acceptable in one era, now are important constraints on protecting biological diversity, maintaining ecological integrity or restoring ecosystem health. These laws also result in potential distributional inequities, both to future generations and across current generations as they favour certain interests over others.

Third, rarely do ecological or social processes coincide with administrative boundaries. Yet, public forest management agencies often are limited by boundaries that cut through watersheds, forests, neighbourhoods and other units (established at one time for good purposes). For example, in Ravalli County, Montana, population growth over the last two decades has been dramatic, rising more than 40% above the level in 1980. Much of this population growth, driven largely by net immigration, has occurred on privately owned forested lands at the urban–wildland interface. Development along publicly administered national forest boundaries has hampered ecological restoration efforts that rely on use of fire, fragmented critical wildlife habitat and challenged the capacity of local, state and federal agencies to protect residences in this fire-prone environment. Administrative boundaries have made influencing and planning for this growth difficult.

Given these economic, policy and administrative insufficiencies and our general lack of knowledge about effects of management at longer time scales and at larger spatial scales, what would we look for in an 'ideal' institutional design?

Paehlke and Torgerson (1990) suggest five characteristics that they argue are necessary for institutions to facilitate informed debate and action on sustainability. First, such institutions must be non-compartmentalized; this means they must reject the functional, isolated structures that characterize natural resource management in favour of structures and processes that facilitate complex analyses, including differing functional areas (e.g. fisheries and forestry) and differing sectoral arenas (e.g. urban planning and natural resources). Second, such institutions are open and thus promote a broadened scale and scope of decision making, replacing largely internal, cloistered processes with early and continuing public scrutiny and access. Third, they are decentralized in order to promote rapid, informed action at the local level. Fourth, they are anti-technocratic. This does not imply a rejection of science or technology, but rather a recasting of their role, changing from one in which they dictate to one in which they inform. Fifth, they are flexible. Flexible organizations reject universal, standardized structures and processes for those with a capacity to adapt to particular conditions and circumstances. As discussed earlier, the growing interest in the concept of adaptive management reflects efforts to foster more flexible structures to deal with the particularistic nature of many environmental issues, and, given the experimental nature of adaptive management, organizations for achieving sustainability will be learning organizations as well.

Developing innovative institutional structures and processes to deal with the challenges represented by the sustainability debate in society is likely the most pressing issue before us. Responding to the complex spatial, temporal and sectoral aspects embedded in the sustainability issue will require heretofore unknown capacities on the part of institutions. As we begin to more carefully formulate the crucial questions sustainability raises, we will gain increased insight about the specific qualities and characteristics required of our institutions, in addition to those noted above. For example, how do we better address inter- or cross-sectoral questions; is it possible to create a strategy for SFM in the presence of unsustainable policies and practices in our cities? Similarly, is it possible for any given nation to put in place SFM strategies when elsewhere, forest management is unsustainable? Is it possible to develop sustainable policies for the future if their present implementation leads to the need for present-day policies that are socially unacceptable? Our demonstrated ability (or inability) to address adequately such questions likely will have profound influences upon our long-term ability to achieve SFM.

4 Selecting indicators of SFM

Against this backdrop of sustainability as a wicked problem and a problematic institutional capacity comes the need to establish indicators of SFM. The interest in indicators is widespread, but one that has involved a number of attempts, ranging from the Liverman *et al.* (1988) discussion of global indicators of sustainability to the Santiago Declaration of C&I (Working Group on Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests, 1995).[4] Each of these implicitly assume some agreement on what forest management should sustain, an assumption we would contend is arguable.

The task here, however, is to identify sub-national indicators that would be useful at the individual forest level. Although sustainability might be a socially acceptable global or national goal, indicators at those levels do little to provide the specific guidance needed in day-to-day decisions of forest managers. For example, an indicator identified in the Santiago Declaration for the criterion biological diversity is the number of forest-dependent species. However, at the sub-national or individual forest level, such an indicator might have little bearing on day-to-day decisions. This indicator may have little to do with sustainability at the local level. As the Declaration itself notes, these indicators ‘are not intended to assess directly sustainability at the forest management unit level’.

Given the conceptual and practical limitations of national level indicators identified above, what are useful criteria for guiding the selection of indicators at the forest level? This question is answered in two parts: first, by understanding the function of indicators in management decisions, and second, by applying tests to ensure that indicators selected are useful and practical.

Indicators serve two important functions: First, through systematic and periodic measurement, indicators inform us of the state of a particular entity, such as a forest. Re-measurement (monitoring) over time reveals changes in forest conditions that might be important to achieving sustainability. Second, indicators let us know how effective particular management actions might have been. For example, an appropriate indicator could be forest structural stage. Management actions at the forest level, such as certain silvicultural techniques, may be initiated to change the distribution of structural stage. Periodic, systematic measurement of structural stage would determine the effectiveness of the management actions.

Thus an important aspect of selecting and monitoring of indicators is an institutional *commitment* and *capacity* to maintain a credible monitoring programme. If monitoring is considered as an afterthought in both management and budgeting, establishing indicators to measure management effectiveness has little beneficial effect. For indicators to serve as more than alarms, there must also be a management strategy in place for what actions to take given certain levels of indicators. Water quality is a good example. When water quality standards are violated, remedial action is taken (such as boil orders combined with system sanitation) to correct the situation.

A second set of questions concerns tests about the characteristics of indicators. To be useful in monitoring conditions and assessing management effectiveness, indicators should reflect several characteristics. First, indicators of sustainability need to focus on the *outputs* of management.[5] Although indicators can reflect certain management inputs (e.g. budget investments), in terms of sustainability the focus is on the *results* of management. Outputs indicate the presence (or absence) of progress toward achieving results.

Thus, an indicator that measures level of expenditure in research and education, such as suggested in the Montreal process, is not relevant because, while it might measure a commitment to research, it does not address the result of research.

Second, indicators must be measurable at the forest level. In this sense, indicators should be quantifiable, not qualitative, in character. Qualitative indicators, such as the degree to which an individual forest may have a commitment to sustainability, are subject to too much interpretation to be useful, and would encourage debate over the meaning of the indicator rather than what it shows. Third, indicators should be subject to reliable measurement; that is, independent observers measuring the same indicators should arrive at the same value. Such a characteristic means that the variance observed is due to true variability rather than measurement error.

Fourth, indicators need to be valid representations of that which they purport to measure. While this would seem to go without saying, it is a particularly relevant question for SFM. Without agreement on what forests *should sustain*, it would be difficult to identify indicators that measure this. For example, should forests sustain ecological integrity or ecosystem health? Indicators of each are likely to be different. Different indicators are needed for different objectives. Any weighting of indicators suggests the relative importance of combinations of objectives and should be the result of interactive processes among the public, scientists and forest managers.

Comparability is a fifth criterion. Comparability across units is important when a standard of acceptability is lacking. In this context, 'standard' is being used in the sense of the 'limit of change' deemed socially acceptable as in a water quality standard, as opposed to a goal to be achieved. Standards are useful for identifying where, when and what types of management actions are needed. Standards for SFM are unlikely to be established in the near future, so an alternative is the comparative method, which relies on uniform indicators. For example, if one forest chooses species diversity as one indicator while another chooses the status of endangered species, then direct comparison is not possible. In many cases, direct comparison is of no interest, and temporal change is more important.

A sixth criterion relates to temporal and spatial scales; the time and spatial intervals across which we attempt to achieve sustainability. Science has a significant instructional role here, but we would suggest, as Dovers and Handmer (1993) have held, that science is limited at larger spatial and temporal scales, affecting its capacity to inform. In a more practical sense, indicators should specify the scales at which they will be measured. This will also help meet the test of comparability.

We argue that selecting indicators that meet these tests is not solely a scientific chore.

Certainly, the criteria involving validity, comparability and scale have strong requirements for substantive stakeholder involvement for public forests; the public is needed to define what should be sustained. Science has important roles in applying these tests and selecting indicators, as we have argued before, but those roles are directed at informing rather than determining what indicators are 'best'.

5 Conclusions

Selecting C&I of sustainability at the forest management unit level is a process conducted within a contentious and socially problematic setting. There is a question about the capacity of current institutions to both select indicators and manage forests to achieve sustainability. Science plays an important role in assisting in the selection and monitoring process, but the various stakeholders have equally meaningful parts.

What appears to be lacking in the selection process are the types of venues Yankelovich (1991) has suggested for dialogue and deliberation, where managers, scientists and stakeholders come together for substantive discussions about sustainability, the value placed on the future and indicators appropriate to those needs. Managers provide experiential knowledge about the biological workings of forests, how forests and people interact and the institutional capacity to manage for sustainability. Scientists can systematically describe cause-and-effect relationships, provide assessments on the limits of formal knowledge, and suggest plausible hypotheses for places where knowledge is limited. Stakeholders come into these venues with preferences and values, a ready set of choices about what should be sustained, and can indicate the social and political acceptability of alternatives.

Thus, the search for SFM is a complex social and technical process, where both stakeholder preferences and scientific information are woven together equally to help achieve politically determined objectives. That this process should be an urgent one should be well understood, for as we procrastinate, we lose options and gain the probability of draconian management actions being implemented. Appreciating complexity, creating appropriate institutional frameworks and equal, interacting involvement of stakeholders, scientists and managers are fundamental prerequisites to SFM.

Notes

1 We acknowledge an alternative perspective which argues that future generations will take care of themselves, because of changes in economic markets, technology and/or public preferences.

2 For example, sustainability deals with a redistribution of power. Power is a systemic characteristic, and its redistribution can only come about through fundamental restructuring.

3 Caldwell also notes that systemic impairment is found in both capitalist as well as socialist economic systems.

4 We recognize that a variety of other national and international processes and protocols concerning C&I of SFM exist.

5 It would be possible to conceive of SFM as an input-oriented activity, with the emphasis on making management sustainable in the sense of the organization maintaining itself. In this sense, one could argue that per acre consumption of energy used in management would be a valid measure of SFM. In the sense we address sustainability here, however, it is a goal or output of management.

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7

SFM Indicators as Tools in Political and Economic Contexts: Actual and Potential Roles

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The situation of society and of forests has considerably changed within recent decades and these changes have some major implications for information requirements. The concept of sustainable forest management (SFM) has gained global political attention as the key to balance preservation and utilization of forests – and its content has been considerably extended to better cover ecological and social aspects. Indicators for a more detailed measurement of and reporting on progress towards SFM have taken a prominent role both in forest policy and in SFM certification in the 1990s. The elaboration of indicators as a tool in forest policy and in business contexts faces many similar challenges, such as the need to specify in concrete terms what is meant by the abstract concept SFM. In both areas people are confronted with methodological weaknesses and a lack of practically useful data in key areas. A comparison of potential application and actual uses of indicators today also reveals that SFM indicators could and possibly will be used in many more areas. Today SFM indicators are still in rather early stages of development. To become a truly useful tool in practice, considerably more work is needed to overcome existing shortcomings and to make indicator sets useful in a broader range of applications.

1 Introduction

A considerable proportion of the world's forests is being used as resource for a livelihood and is or has been under some form of management. Forests often played an important role in a nation's wealth and they fulfil a broad variety of functions, from the utilization of products to providing services such as recreation, preserving biodiversity, cultural and spiritual values, or for basic life support systems, such as water and air cycles. The importance of forests and how society organizes their utilization differs considerably throughout the world. The core concept of preserving forests while at the same time utilizing its benefits, namely SFM, has century-old roots. However, over time the contents of this concept have changed considerably, adapting to new knowledge and to new needs.

More recently the increase of global population, industrialization and a more systematic utilization of forests in many areas of the world have caused considerable changes of the situation of forests and have triggered fears of different kinds. A good part of society perceives these changes as problematic. Environmental pressure groups have evolved and point to the possible negative consequences of current practices. In recent years initiatives increasingly try to combat negative consequences of ongoing environmental changes and to foster efficient and effective resource use. Probably the most relevant of these efforts are those defining a common global political goal: sustainable development. In forestry this has resulted in defining or redefining the meaning of what constitutes SFM as well as elaborating criteria and indicators for its measurement.

The objective of the chapter is to review the application of indicators for measuring aspects of SFM and the further use of indicators in two societal sub-systems: the economic and the political systems (Fig. 7.1). The chapter tries to answer how indicators used in forest policy contexts might affect forest management units (FMUs) and how indicators can be usefully applied by FMUs in business contexts.

The chapter sets out the potential use of SFM indicators in forest policy and as tools in business or market systems. It shows the actual use of indicators in various ongoing processes and their likely future importance, and discusses the weaknesses of SFM indicators that exist today. The focus of the contribution is on their application in temperate zones, especially Europe.

2 SFM and indicators

2.1 Definitions and background

SFM is a normative concept whereby society defines broadly which aspects of forests, their components, processes or functions are to be preserved over the

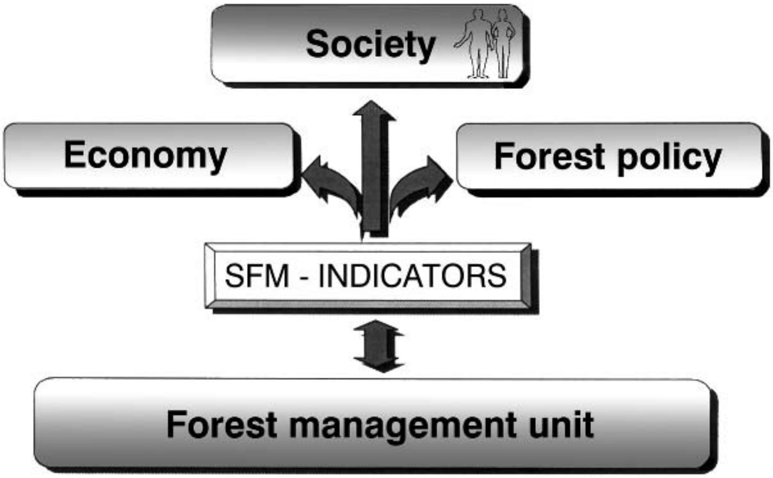

Fig. 7.1. SFM indicators provide a link between the FMU and societal systems.

long term and which are to be utilized. The necessary political judgements are mainly based on values and knowledge. Values and knowledge vary throughout the world and over time. SFM is therefore a dynamic concept, the result of ongoing political processes driven by the various actors and their respective values, interests, knowledge and relative negotiating power.

For a long time, 'sustainability' was almost exclusively concerned with sustained yield of wood. Over the last century one can observe a steady increase in the importance of other, previously little recognized aspects of forests, such as ecological and social functions. In the last couple of years two major issues brought important changes: the contents SFM were in general broadened, ecological as well as socio-cultural aspects were strengthened, and SFM became a central issue globally. Today, the concept of SFM rests firmly on three pillars (Fig. 7.2).

A first global non-binding definition or description of the contents of SFM can be found in the United Nations Conference on Environment and Development (UNCED) 'Forest Principles', signed in 1992. Principle 2(b) reads: 'Forest resources and forest land should be sustainably managed to meet the social, economic, ecological, cultural and spiritual human needs of present and future generations'.[1]

The development of criteria and indicators (C&I) to further specify abstract definitions of SFM, to measure and report on the state of the art and changes related to SFM, gained one of the highest priorities in forest policy globally. As the word 'indicator' already assumes, indicators are used to point out something – to inform about an aspect by making use of empirical data. They are primarily applied in areas and for concepts that cannot be fully operationalized. Probably the most often used working definition for indicators

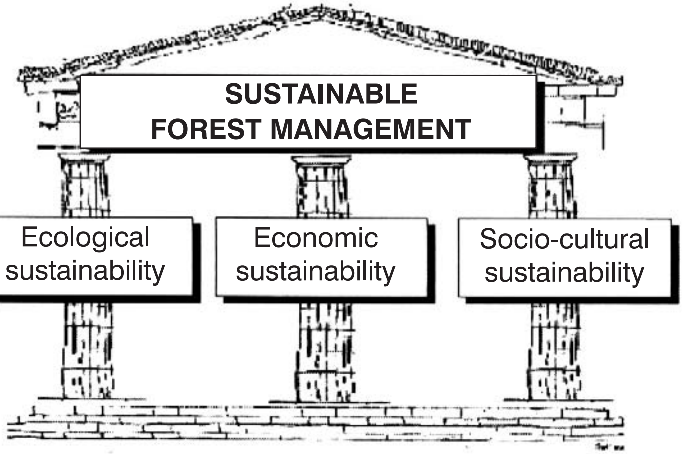

Fig. 7.2. The three pillars of contemporary SFM.

in ongoing processes is: 'indicators act as signs or symptoms for the presence of something'. The role of indicators is thus to:

- simplify complex issues;
- show the status at a certain point in time (single measurement);
- show changes over time (multiple measurement = monitoring).

The main determining factor for the content of an indicator is the information interest of the user. In the world of politics or in business, indicators are often used as decision support information in management processes, including monitoring of the external environment, planning, implementation and evaluation of activities. Examples of widely applied indicators in the political as well as in the economic world are the rate of inflation, gross national product and unemployment rate. If they are to be a useful tool, indicators must be relevant, reliable, valid and easy to measure. Indicators vary in:

Type
- type of information (e.g. factual, prognostic, conjunctive, normative);
- type of data (quantitative – qualitative);
- aggregation type (single parameter – composite index).

Scale
- geographical scale (e.g.: global – national – FMU);
- time scale (one-shot – continuous; short term – long term).

Indicators can both be quantitative and qualitative. For each quantitative parameter of an indicator, a specific unit of measurement (e.g. pH, units m^{-2}, ha) has to be given. Information is collected 'on the ground' and condensed to the level required for the specific application (Fig. 7.3). Depending on the sort of information required, indicators are applied on a variety of aggregation levels.

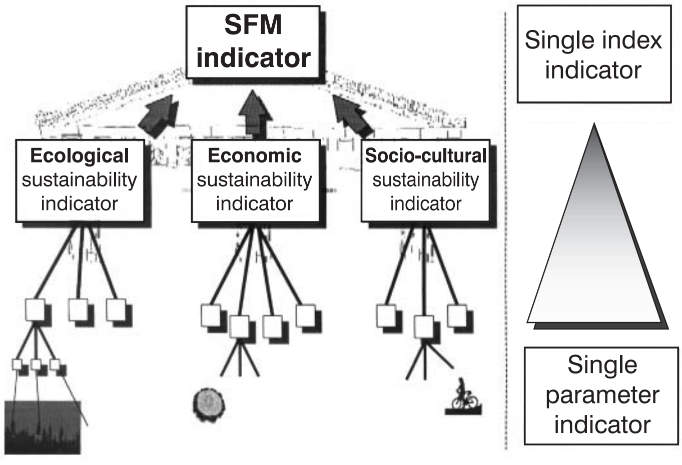

Fig. 7.3. Types of indicators to measure aspects of SFM.

Some, like Ott (1978), define indicators also on an aggregate scale and use the term identically with 'index',[2] others link an indicator to one specific type of data. Any aggregation of different types of data has to be based on a defined method of aggregation, including weighting of the parameters used for the composite index.

Which aggregation level to choose for an indicator depends on the user/target group/receiver of the information (Vos *et al.*, 1985). Indicators for the general public require a rather high level of aggregation, indicators for policy makers require a medium level of aggregation and indicators for forest managers require little aggregation.

Numerous institutions, at levels ranging from international to local, currently make extensive use of indicators in information systems on environmental issues. These include UN organizations, especially United Nations Department for Policy Coordination and Sustainable Development (DP-CSD), the Organisation for Economic Cooperation and Development (OECD), the World Bank, International Union for the Conservation of Nature (IUCN), national governments, business enterprises and others.

2.2 Indicators for measuring aspects of sustainable forest management

Many of the ecological, economic and social aspects of forests and their management have been measured and assessed in a variety of contexts and for a multitude of uses. These include policy and planning by the forest owner, governmental forestry departments or by other organizations. Forest owners in many regions have used indicators such as 'growth : removal ratio' (a composite-index indicator) or 'area under forest cover' (a single-parameter indicator) to plan, execute and evaluate their operations. In the field of forestry, the much-increased recent efforts to develop indicators are almost exclusively connected with the measurement and external reporting of the degree of sustainability of forests and/or forest management.

The conceptual framework that is often used to define SFM in Europe and elsewhere is based on the notion, broadly accepted in international political fora, that SFM includes economic, ecological and social components. In order to operationalize abstract definitions of sustainable forest management within the framework of economic, ecological and social sustainability, a hierarchical system of 'principles/standards, guidelines, criteria and indicators' has often been adopted. In a policy cycle, a specific role can be assigned to each of these different tools (Fig. 7.4).

Principles are used to lay down general objectives at an abstract level.[3] Standards are prescriptions of quality requirements. They are most useful in connection with the specification of related threshold levels against which an evaluation can take place. Guidelines are used to formulate actions that should ensure the achievement of the overall goal of SFM, as laid down by principles or standards. Guidelines can be voluntary or obligatory. Criteria are used as a tool to evaluate the attainment of a specific objective related to SFM.[4] A common definition and consistent use of the term 'criterion' is lacking, which creates problems in communication between different parties.[5] Indicators, as has been described, are used to show the state of the art and monitor changes in relevant aspects.

Although the elements or instruments described in Fig. 7.4 are not applied consistently by the various actors within the different political and economic

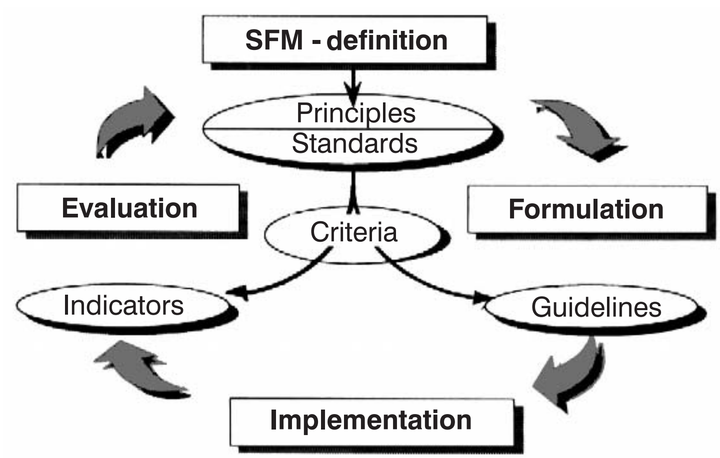

Fig. 7.4. Methodical approach to operationalize SFM definitions in different contexts.

fora, they can be frequently found in most of the initiatives. They include, for example, environmental management systems in economic areas and governmental programmes in forest policy areas.

C&I are in effect elements that clarify what is meant by SFM in practice. SFM has furthermore to be measured on different scales, e.g. from global, geo-regional, national or sub-national, to FMU scale.[6]

The broadening of the definition of SFM has considerable impact on the indicator frameworks to be used. As new definitions of SFM strengthen ecological and socio-cultural aspects of SFM while retaining the economic aspects, new indicators are needed to cover the expanded areas adequately.

SFM indicators simplify and help to effectively organize and perform a two-step process (Table 7.1): collection of data and its subsequent utilization as information.

3 Forest policy and the use of SFM indicators

3.1 The rise of SFM indicators to centre stage in forest policy

The last three decades of the 20th century saw a rising threat to the quality and quantity of forests on a global scale. In the tropics, the rate of deforestation was drastically increasing, resulting in concern about the large losses of biological diversity, the increasing amounts of carbon dioxide released into the atmosphere, the fate of indigenous forest dwellers and other consequences. In Europe, forests faced considerable threats of declining health and vitality. As a result, public and political interest in and concern about the situation of forests increased throughout the world.

Table 7.1. SFM indicators allow effective transformation of data into information.

Step 1: Collection of data

- Status of forests:
  forest components and/or structure (e.g. biodiversity, soil, water)
  processes (e.g. health and vitality)
- Impact on and output from forests:
  impacts on forests from forestry or society
  output from forests (products) or other outcomes or effects
- Monitoring of changes

Step 2: Use of information

- Problem recognition
- Objective determination
- Formulation of strategies
- Implementation
- Evaluation

One factor that was common to the majority of the perceived problems was their international scale. Deforestation took place in the majority of countries in the tropics because of population pressure, industrial exploitation or other, usually bundles, of causes. Likewise, the sources that caused acid rain in Europe were often found outside the sphere of influence of governments where damage occurred. The policy arena that was seen as being able to address the issue appropriately was international.

In the resulting political process of setting up global or regional international regimes, the concept of SFM was for the first time broadly discussed on a global level. The most important international policy arena has been the UNCED, held in Rio de Janeiro in 1992, and its follow-up processes. In this political environment, 'sustainable development' (SD) was regarded as the most suitable concept to solve apparent problems, and SD was adopted as a common global political goal. The formulation of the general approach was comparatively easy, however, compared to the next step: to make clear what sustainable development means in practice and to implement it. Fortunately, work to define and operationalize SFM on an international political scale by making use of criteria and indicators had already begun in the forestry sector before the UNCED conference.

The first international body to develop a set of criteria and indicators for SFM was the International Tropical Timber Organization (ITTO), who presented a list of five criteria and 27 'possible indicators' in 1992 (Table 7.2).

A Seminar of Experts on the Sustainable Development of Temperate and Boreal Forests was held in Montreal in September 1993 under the aegis of the Conference of Security and Co-operation in Europe. This scientific and technical forum examined the scientific basis for the concept of conservation, management and development of temperate and boreal forests. The major outcome of the seminar was a preliminary set of criteria and some potential

Table 7.2. Forest policy and indicators: the key processes in development of SFM indicators.

1992	ITTO – Criteria and Indicators
	UNCED – Forest Principles and Agenda 21
1993	CSCE-Seminar
	ATO Principles, Criteria and Indicators
1994	'Pan-European C&I' (MCPFE/'Helsinki C&I')
1995	'Montreal C&I' (Montreal Process)
	Tarapoto C&I
	CSD-IPF Topic
	FAO/UNEP Dry Zone Africa
1996	FAO/UNEP Near East
1997	Lepaterique - Central America
1999	Dry Forests in Asia

indicators for SFM. This set served as an indicative guide and as background material for several other initiatives.

In Europe, the follow-up process for the Resolutions H1 and H2 of the Second Ministerial Conference on the Protection of Forests in Europe (MCPFE) in July 1993 also started to consider C&I for evaluation and reporting in 1993. Subsequently, a set of C&I was prepared. In 1994, six criteria and 27 'most suitable quantitative indicators' to be applied at a national level were adopted by the national representatives of the signatory states of the 'Helsinki Resolutions'. In 1995, 101 'descriptive' indicators for possible use were added to the list. The 'Pan-European C&I' is the only set of C&I worldwide that is supported by political commitment at ministerial level by the participating governments of the MCPFE.

The 'Working Group on Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests', subsequently called the 'Montreal Process', was formed in 1994 to advance the development of internationally agreed C&I, at the national level, for the conservation and sustainable management of temperate and boreal forests in regions outside Europe. In February 1995, the countries represented in this group endorsed a comprehensive set of seven criteria and 67 associated indicators for forest conservation and sustainable management for use by their respective policy makers.

Senior government officials and experts of the Amazonian countries produced a document containing a report of C&I for measuring the sustainability of the Amazonian forest in February 1995 ('Tarapoto Proposal'). The Tarapoto Proposal consists of 12 criteria and 77 indicators (national level: seven criteria and 47 indicators; management unit level: four criteria and 23 indicators, and services at global level: one criterion with seven indicators). It is intended to function as guide for the countries which are signatories to the Amazonian Cooperation Treaty.

FAO and the United Nations Environmental Programme (UNEP) helped to establish regional initiatives to elaborate C&I in regions that were not well covered by the Helsinki, Montreal or Tarapoto initiatives. A UNEP/Food and Agriculture Organization of the United Nations (FAO) Expert Meeting on Criteria and Indicators for Sustainable Forest Management in Dry-Zone Africa was jointly organized in November 1995 and prepared a set of seven criteria and 47 indicators to be applied on a regional scale. In the Central American process ('Lepateric Process') four criteria and 40 indicators were proposed for application at a regional level, while at national level eight criteria and 52 indicators were proposed.

The implementation of the UNCED Agenda 21 is administered by the UN Commission on Sustainable Development (UN CSD). The CSD established an Ad-hoc Intergovernmental Panel on Forests (IPF) in 1995 as a broad political forum that focuses on priority issues. One of the five priority issues of the IPF was 'Scientific research, forest assessment and development of criteria and indicators for SFM'.

Table 7.3 gives a summary of ongoing international initiatives on development and implementation of criteria and indicators for SFM.

Table 7.4 lists the contents of the criteria of the Pan-European and the Montreal process. One can see that the contents of criteria sets are very similar between international forest policy processes. These criteria build the framework for indicators which are designed to deliver information on each of the aspects listed.

Table 7.3. Coverage of ongoing initiatives on SFM criteria and indicators by ecological region (UNCSD-IPF 1996; own data).

C&I initiative and ecological region	No. of countries	Indicator level	No. of C&I	
			Criteria	Indicators
Temperate and boreal forests				
Pan-European Process[a]	38	National	6	27/101
Montreal Process	12	National	7	67
Tropical forests				
ITTO Producer Countries	25	National + FMU	5	27
ATO Member Countries	13	National + Reg	28	60
Tarapoto Proposal	8	National + FMU	12	77
Central America	7	National +	8	52
		Regional	4	40
Dry-zone forests				
Sub-Saharan Dry-Zone Africa	27	National	7	47
Near East	20	National	7	67
Dry Forests Asia	9	National	8	49

[a]The number of countries for Helsinki refers to the signatory states of the Helsinki Resolutions as well as those countries who have subsequently participated in the work of the 'Pan-European Forest Process'.

Table 7.4. Indicator framework of forest policy processes for the temperate and boreal zone.

Pan-European C&I Process	Montreal Process
1. Forest resources and contribution to global carbon cycles	1. Biological diversity
2. Forest ecosystem health and vitality	2. Productive capacity
3. Productive functions (wood, non-wood)	3. Ecosystem health and vitality
4. Biological diversity	4. Soil and water resources
5. Protective functions in forest management (notably soil and water)	5. Contribution to global carbon cycles
6. Other socio-economic functions and conditions	6. Multiple socio-economic benefits
	7. Legal and institutional framework

Several comparisons of the contents of the C&I of these initiatives are available, including the Intergovernmental Seminar on Criteria and Indicators for Sustainable Forest Management (ISCI) (1996), Bueren and Blom (1997), FAO (1996, 1997) and International Standards Organization (ISO)/TC207 (1997).

3.2 Potential uses of SFM indicators in forest policy that might affect FMUs

In theory, the main purpose of indicators for governments is to get more and better information on 'reality' regarding the condition of forests and their utilization, and to use this information for the design of effective and efficient policy interventions.

In general, SFM indicators can be used in forest policy in two areas (Fig. 7.5).

Collection of information

The collection of information on the status of forest components, such as biodiversity, soil, water, processes, as well as on impacts on and outputs from forests is of course is not a new thing; governmental forest inventory services are well established institutions in many countries. What is new is that the contents of current inventory as well as methods are under political discussion

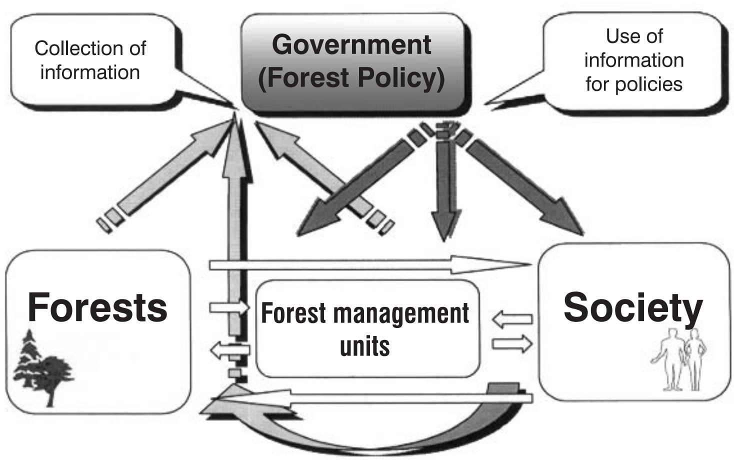

Fig. 7.5. The use of indicators in forest policy: data collection and information utilization.

and under pressure to determine whether or not to update inventory systems and what new indicators to include in possible revisions.

The collection of information on SFM has to focus on:

1. Three systems
- Forest ecosystem
- Forest management system, i.e. the economic system
- Society

2. Three interfaces
- Forest management – forest ecosystem (direct impact, output, secondary effects)
- Forest management – society (direct impact, output, secondary effects)
- Society – forest ecosystem (direct impact, direct effects, secondary effects).

The information based on the actual status quo of forests is often weaker than the general society would expect. Even for very basic indicators like existing global forest area and changes in area, the figures differed considerably between different sources even in the late 1980s.[7] This situation can to a certain degree be attributed to different underlying definitions and data collection methods. However, a good part still seems to be caused simply by lack of accurate data. Mather wrote in 1990 (s. 58): 'Since the early 1970s, our knowledge may have improved to some extent, but many uncertainties remain. . .. Even during the last two decades, estimates of the forest area have varied from under 3000 to over 6000 million hectares, or from 25 to 40 percent of the global surface'. The situation has greatly improved in the last decade due to the increased international political interest in forests. However, the commonplace 'We know a lot about the moon, but we know little about our world's forests' is still true if it comes to biodiversity and related functional interactions.

As a matter of fact, information is lacking on exactly those aspects that constitute the hottest political issues, such as biodiversity. More information is available on other forest components like soil or water. Very little information is available on the actual and historical impacts of forest management, hidden behind discussions on the 'naturalness' of non-virgin forests, 'potential natural vegetation', and 'hemeroby' (the effect of human activity). Considerable difficulties also exist in measuring non-marketable or non-marketed output or effects of forests that benefit society ('passive use values'). In some cases it is not a lack of information but the fragmented and very detailed and local nature of existing information that makes its direct use impossible.

With regard to the collection of information, the hot issues in forest policy in temperate zones are:

1. Ecological
- Biodiversity
- Protection/conservation

2. Economic
    - Non-wood forest products and services marketing
    - Non-use values of forests for society
3. Socio-cultural
    - Recreation
    - Participation.

Use of information

The possible uses of information on SFM that is collected through indicators is of considerable political concern for the parties most directly affected, namely private forest owners. The range of the possible user groups is vast, but only some are able to use the information directly for political activity.

The core user groups of information on SFM indicators collected in forest policy contexts are governmental organizations, such as forest policy institutions, environmental institutions or national accounting services, forest owner and forest owner interest groups, and environmental groups.

These groups can use indicators at a national or regional level as well as at the FMU level to identify potentially problematic areas and for formulating, implementing and evaluating related policy programmes.

In relation to governmental organizations, information from SFM indicators can feed into policy action that uses one or more of the following instruments that affect FMUs. Indicators can play a role in implementing forest policy through:

1. Regulatory instruments
2. Economic instruments
    - incentives
    - disincentives
3. Informational instruments

Regarding regulatory instruments, indicators can be used as a reference for forest laws and regulations. On a FMU-level, they could be used to evaluate adherence to laws and other regulations. For economic instruments, indicators on a FMU level can be used as a tool to distribute economic incentives and disincentives. Indicators on a regional or national level can be used to identify regions for specific subsidy programmes or for funding specific projects. Indicators have a wide range of potential uses as information instruments, and especially as communication tools for political actors, foresters and the general public. They provide tools for monitoring and assessment by foresters, assistance in or development of certification programmes, and information of the general public and others. In the context of sovereign regions or countries, indicators can be used to compare similarities and differences in the respective situations and approaches.

3.3 Actual and potential further uses of SFM indicators in forest policy

The development of indicators for measuring SFM in forest policy has mainly taken place at an international political level. The majority of these initiatives has explicitly or implicitly excluded FMU-level indicator development. Exceptions are the ITTO, the Tarapoto group and the Central America initiative. An important factor influencing whether or not indicators for FMUs have been developed is the existing forest ownership structure. Private forest owners usually are against giving away detailed information on their enterprises.

The selection and development of the C&I is, in all processes, strongly influenced by values, interests and power of the participating negotiating parties. Several issues were, are and will be subject to intense 'hidden bargaining' between and within different institutions and interest groups to maximize the formal and informal interests of each group in the process of specifying actual changes in data collection and use on international, national and subnational levels. The outcome of these negotiating processes largely determines the effectiveness and efficiency of the use of SFM indicators by politicians and thus the impact of SFM indicators on FMUs. The main issues of political debate on indicators concern:

- What information will be collected
- Who will collect information
- How will information be used
- Who will use (or be able to use) the information collected.

The Pan-European Process on C&I focused on devising a commonly agreed indicator set at the national level that was practical in having few or no additional resource requirements. New resources for monitoring systems in Europe, or new concepts and systems adjusted to meet new information requirements on SFM, were not the main aims, nor was the harmonization of methods for the collection of data by the existing information systems a central theme. Extensive use is made of existing data from national inventory systems employing varying collection methods. The actual use of indicators in the Pan-European Process focuses primarily on better sharing of existing information at the national level within political actors in forest policy in Europe, and only to a marginal extent on devising consistent strategies to reach the general public. The development and selection phase for C&I also had an important function in building awareness regarding the changing content of the concept SFM.

The actual use of indicators in the Montreal Process is to a good part characterized by building awareness about the characteristics of SFM, as well as the collective identification of gaps in information, be it in knowledge, databases, monitoring or others (Montreal Process, 1997). The political commitment of participating countries is far less strong for this process, allowing a more technical–scientific approach to the development of a set of C&I.

Indicators for SFM developed in the context of forest policy have up to now mainly focused on the use of indicators as informational instruments within the forest policy community. However, the processes of devising C&I systems to collect information on SFM are not yet concluded. ITTO has recently reviewed their set of indicators, and the Pan-European C&I are likely to be reviewed to better accommodate biodiversity and socio-cultural issues. The 'Montreal C&I' are also likely to undergo further changes over time.

Apart from serving as an information instrument within the community, further uses of indicators have already been explored. However, this is only just beginning. Today the use of SFM indicators is strongly dominated by two issues: national reporting of SFM by governments in forest policy, and forest or timber certification in business applications.

In the future, indicators will be an important tool to assess existing forest policy and to adapt national or regional forest programmes to new requirements. The range of actual uses of indicators that link forest policy with the forest management unit level is shown in Table 7.5.

The main application of C&I today is in communication on SFM within the forest policy community itself. Communication between foresters, using C&I sets, is taking shape within the Pan-European Process with the adoption of common voluntary 'Pan-European Operational Level Guidelines' that are applicable at the FMU level, and through other international and national initiatives. Several national bodies have taken initiatives to develop national or sub-national indicators or voluntary standards, referring to existing indicator sets, for application at the FMU level. An important indirect effect of existing indicator sets that has already been mentioned is their awareness-building effect at all levels. This effect has played an important role in information diffusion, both in national and international processes.

Communication to the general public on the basis of C&I results is still in its infancy. In Europe, several public opinion polls were conducted recently to get an understanding of society's views on forests. Subsequently forestry

Table 7.5. Actual and likely future uses of SFM indicators in forest policy. The number of stars denotes the frequency of application.

Potential uses of SFM indicators	Application today	Likely future use
Regulatory instruments		**
Economic instruments		
incentives	*	***
disincentives		*
Informational instruments		
to forest policy	******	******
to foresters	*	****
to general public	*	****

communities in some countries started public campaigns to counter common misconceptions of the general public, such as that of diminishing forest areas.

C&I can play an important role as reference for international, national and sub-national policy formulation regarding regulatory instruments, such as for a potential future international legally binding instrument on forests. Indicators developed in international processes can be and have to some extent been used as a loose or clear reference for evaluating contents of existing national or regional legislation. However, indicators were not expressly designed for this purpose and have not yet been directly applied in this context. Indicator sets might also be used by governments to evaluate and regulate the various emerging private certification schemes.

Subsidy systems based on evaluation of SFM by using indicators have been developed and are being tested, e.g. in the 'Waldökopunktesystem' (Forest-eco-points-system) in some regions in Austria. Similar systems are used for the allocation of project funds by international donor organizations. This area of political application of SFM indicators has the potential to become the second major field of application, after their use as a communication tool.

In the context of the general political aim of achieving a sustainable society, forestry is advanced in its conceptual homework. A survey in September 1997 of an expert panel on sustainable development revealed that the forest industry sector is seen as managing its transition to sustainable development better than the rest of 11 other sectors (Environics, 1998). However, higher aggregation of SFM indicators is necessary to communicate progress more effectively. It would also be of advantage to have a voice in designing economy-wide policies aimed at sustainability.

4 The use of SFM indicators in business contexts

4.1 A development that got worldwide attention: certification

Technological and economic changes of the last century led to a shift from an agricultural society through an industrial phase to a service-oriented society in the western world today. These changes in the macro-economic context diminished the importance of the forestry sector. It has become a marginal contributor to national income in the majority of countries today, and a fairly small minority of the population in temperate areas depends mainly on income from forestry. The changes also led to socio-demographic developments that had far-reaching consequences for the relationship between mankind and nature.

Intensive resource use, emerging waste disposal problems, urbanization, increased disposable income of a broad stratum of the western population, media influence, tourism and other factors have led to psychological shifts in the perception of the environment by society in general, and of forests and its perceived role in particular. Interest groups that aim to make people aware of environmental changes and to improve the situation of the environment became prominent in the course of these changes. Many of the most prominent issues, such as biodiversity, tropical rainforest or acid rain, have strong relations to forests generally.

One of the most recent global initiatives by environmental groups to arrest the ever-deteriorating situation of forests worldwide is certification of 'well managed' or 'sustainably managed' forests. The certification of sustainably managed forests and the subsequent labelling of sustainably produced timber was first raised in debate by non-profit organizations at the end of the 1980s. Since then a multitude of certification initiatives have evolved worldwide, resulting in a proliferation of definitions and certification standards, including C&I for assessment of these standards. The use of standards, however, requires the definition of threshold levels for judgement and thus a clear definition of what is good and what is bad.

Certification as such is a purely market-driven approach to communicate high quality levels. Certification is based on assessment (by an independent external party) whether the aspect in question meets certain quality requirements, called 'standards'. These standards can in principle be set by anyone and on any quality level.

In 1993 a Forest Stewardship Council (FSC) was established in an attempt to harmonize certification schemes and establish a global framework of performance standards, namely principles and criteria for SFM, and to act as an accreditation body for certification organizations. The FSC mainly has an environmental and social non-governmental organization background, and is the only existing global body that tries to ensure the credibility of product claims of 'well-managed forest'.

In 1995 the ISO was brought into the debate by the industry to act as a standard-setting body for specific sector process standards on sustainable forest management within the ISO 14.000 series on environmental management systems. Subsequently work started on relevant standards. However, as specific sector standards are not foreseen within the series, a bridging document has been prepared by a working group within the responsible Technical Committee (TC 207) that offers voluntary guidance for forestry organizations in the use of ISO 14.000 series implementation in forestry (ISO, 1998).

In 1998 the Pan-European Forest Certification (PEFC) Initiative was launched by national forest owner and forest sector interest groups of several European countries. PEFC acts as an umbrella body for individual national certification systems and is intended to better accommodate the needs and requirements of the many forest owners in Europe with small forest estates. The standards used by PEFC are based on the pan-European criteria and indicators and the pan-European operational level guidelines adopted by the Ministerial Conference on the Protection of Forests in Europe.

The assessment of the sustainability of forest management is usually based on indicator frameworks. Aspects of harmonization/standardization of these SFM indicator frameworks and the potential and limits for standardized measurement has up to now seen efforts by the Center for International Forestry Research (CIFOR) (Prabhu *et al.*, 1996, 1999) and FAO (1996).

4.2 Potential uses of SFM indicators in business contexts

The dominant actual use of C&I today is in certifying sustainably or well-managed forests. However, various other uses of SFM indicators exist. In general, two broad fields of uses for SFM indicators at the forest management unit level can be recognized:[8]

1. Indicators for internal use as a basis for decisions by forest managers about
- the forest estate and forest management
- markets
- governmental bodies and peer groups.

2. Indicators for external use as information and communication tools to
- the general public
- clients, customers, (products and services/input and output, e.g. insurance)
- peer groups
- governmental bodies.

Indicators for internal use by forest managers, in the form of decision support systems or market information systems, are as old as forest management itself. Several of the indicators used today have long been used by forest managers, although they were not previously called 'indicators'. Information on the forests managed, forest management and its effect on the forest are to a varying degree part of any forest management system.

SFM indicator frameworks, however, do not only have to comprise internal information on the forest and its management, but also information on the market for products and services as well as on the major forces influencing economic sustainability, such as government, environmental groups and local communities.

Indicators for the forest estate and forest management are usually well established. However, there is a requirement to update or adjust existing SFM information systems to incorporate information on those issues that have recently shifted into focus in society at large and in policy, namely ecological and social–cultural considerations. These changes in the general framework for business seem to have barely influenced the information systems of companies. The costs involved seem to outweigh by far the benefits expected by forest managers from such exercises.

The collection of market information, comprising customers and competition, is daily business for companies. Forestry, as provider of the raw material, wood, and far from the end user, has been quite passive and reluctant to react to changes in society and to collect market information in the services area. Activity of this sort, however, is clearly part of an SFM indicator-information system. Collection and integration of data on socio-cultural aspects of the non-market interface between forestry and society provides information related to risk management, including issues for local communities, environmental groups and others.

External communication on SFM seems to have been a widely neglected field in forestry, despite clear signals by general society. The performance of a company in ecological, economic and social terms is of potential interest to many, ranging from clients, consumers, government, environmental groups to insurance companies, other peer groups and general society. The usefulness of external communication about the activity of a company is only recently beginning to be seen as an important aspect of 'brand building' or risk reduction. Today, indicators are being increasingly used by companies to produce information for use in various forms of external communication.

The information tools to be used have to be tailored to the interests of the target audience and communication channels available. Indicators have an expanding role in all sorts of 'green' communication, environmental reporting and the various product or process certification schemes or forest registration schemes that have emerged in recent years (ISO 14.000 series, FSC, eco-labels, etc.) or are currently being developed.[9]

Certification has three major communication functions:

- to reduce risk
- to enhance credibility
- to improve the image.

Different end-user groups, be they the general public, politicians, foresters or others clearly have different information needs. In order to communicate effectively, it is necessary to understand beliefs and attitudes of society about the subject in question. In regard to the general society in Europe, several public opinion polls have shown that people still believe, contrary to the facts, that forest areas are diminishing in their home countries. The polls also showed that people find preservation issues (especially biodiversity) and services more important than ensuring wood supply (Rametsteiner, 1999).

Moreover, more than 10 years after 'sustainable development' was adopted on a global political level as the leading concept for the coming century, only a fraction of the population has ever heard of the idea, let alone understood it. This is the case even in a country such as Germany, which is regarded as being in the forefront of environmental awareness (Preisendörfer, 1996). The same is true for the concept of SFM. This potentially strong competitive advantage of the forestry sector, namely its theoretical ability to come at least very close to sustainability, is currently not used as a competitive sector advantage in communication. If indicators are to be chosen to address these aspects, indicators or indicator sets have to be specifically selected and adapted in terms of content and level of aggregation. To highlight the importance of the passive use values of forests, it would also be advantageous to have indicators at the FMU level for communicating these values.

5 Discussion

The situation of society and of forests has considerably changed within recent decades, and these changes have major implications for information requirements, both internally and for external purposes. Information requirements have changed both quantitatively to include more aspects, such as protection, biodiversity and social issues, and qualitatively to include information on forest quality and other properties. However, a lack of awareness regarding the integration of new requirements and efforts to communicate SFM to the general public can be observed both in forest policy fora and in forest management. There is also a problem with positive results of external communication, as costs have to be borne by individual agents, but benefits can largely be shared by the whole sector.

Indicators are potentially a useful tool for conveying information about complex matters in a simple manner. Indicators have a considerable range of potential uses, both in forest policy and for forest management. Today, this range of uses is overshadowed by two prominent applications: national reporting within international initiatives in forest policy, and for certification. For most of the other potential uses, the constraints as well as costs and benefits involved in the practical application of SFM indicators have not yet been sufficiently explored to enable conclusions to be made about the limits of utility. Neither has the necessity of adapting indicators for specific applications been studied in great detail. Some of the most salient issues that need considerably more input are as follows.

5.1 Lack of data, adequate indicators and threshold levels

In several areas, the practical use of indicators as tools is considerably constrained by the sheer lack of data or adequate (relevant, reliable, valid and cost-effective to use) indicators. This fact is a result of both a lack of in-depth knowledge about some objects of measurement and the difficulty of conveying accurate information on some aspects through indicators. Some of the most important research areas to be addressed are biodiversity and the evaluation of non-marketed effects of forests.

Multiple approaches might better address information requirements in some areas. The value of combining different types of indicators, qualitative and quantitative, and other instruments of evaluation, e.g. policy evaluation, should be explored. Considerable further research is needed to develop evaluation tools that can be more easily used by different non-scientific end-users, and to investigate the limits of practical use of indicators.

A further key area lacking adequate information is that of threshold levels. Some of these can be determined by natural sciences, but for a good part this determination requires social negotiation processes. Such ‘performance standards’ are very difficult to define for wider geographical areas.

5.2 Methodical weaknesses related to SFM indicators

Definition of key terms and concepts

The majority of terms used are not, or not sufficiently, defined. This applies both to the conceptual framework (criteria, FMU) and its contents (forest, biodiversity, health and vitality, etc.). Even accurate measurements by different evaluating organizations are thus difficult to compare. There are also very different conceptual approaches with C&I sets in use, whose commonalities and differences have not been adequately explored yet.

Measurement methods

The methodical prescription of data measurement (units of measurement, spatial resolution, time scale/continuity, aggregation) is often weakly defined and/or is subject to change. This sharply reduces comparability and the meaning of comparisons of results from different measurements, either by different users or over time. Furthermore the combined use of a multitude of methods for measurement, such as geographic information system, remote sensing and others, should be further explored.

Evaluation methods

Methods of evaluating measurement results, especially procedures for weighting and aggregation as well as for determining threshold levels for assessment of the degree of SFM, are often not sufficiently clearly defined, leading to wide variability in interpretations. In the context of certification, this leads to the question as to how far claims made are justifiable. International harmonization and standardization of performance standards, and thus comparability of results, is achievable only on a very general basis.

Harmonization and standardization of methods

Lack of harmonization and standardization of definitions and methods between countries and initiatives also sharply reduces their comparability and the meaning of comparisons of results. The theoretical possibility of standardization and harmonization of contents of measurement is already constrained by the complexity of the subject matter. Measurement of very different complex living systems and effects of interactions on the one hand, and the goal of measuring the complex normative abstract concept of SFM on the other, is far from easy.

It is also obvious that international forest policy processes on indicators cannot replace processes at the national, sub-national or management unit level to develop and refine indicator systems and frameworks for use at the sub-national or FMU level. However, C&I sets developed in international forest policy processes do provide a valuable starting base and a framework for more detailed and user-adapted indicator sets.

Judgements of absolute sustainability are unlikely to be ever possible, due to imperfect knowledge and changing values in society. What can be measured, however, is relative sustainability: whether society is moving away from or towards sustainability. This requires the aggregation of a broad variety of single-parameter indicators to form a composite index. Due to the normative nature of the SFM concept, the necessary weighting of the single indicators will again require broad consultation and negotiation with ‘all stakeholders’.

5.3 Low awareness of the multitude of uses

The third area for further investigation concerns the possibly useful but as yet unexplored applications of the tool C&I in a multitude of contexts, on international, national and sub-national levels.

6 Conclusions

In a changing world information requirements necessarily also change. Indicators as an information tool have attracted widespread attention in political and other fora within a short period of time. Indicator systems to measure SFM are still in their early stages of development, and they are not yet fully applied. Their potential of becoming a major and useful tool to more fully understand and implement SFM by foresters, politicians and others is clear. However, the extent to which SFM indicators become a practical tool is to a good part dependent of further work. The practical constraints, including cost and benefits, of widely applying SFM indicator frameworks have to be further explored.

Notes

1 A/Conf. 151/6/Rev. 1 of 13 June 1992 of the 'Non-Legally Binding Authoritative Statement of Principles for a Global Consensus on the Management, Conservation and Sustainable Development of All Types of Forests' ('Forest Principles').
2 'Ideally, an index or indicator is a means devised to reduce a large quantity of data down to its simplest form, retaining essential meaning for the questions that are being asked of the data. In short, an index is designed to simplify'; Ott (1978).
3 A widely used definition of a principle is 'a fundamental truth or law as the basis of reasoning or action'.
4 Criteria are commonly defined as 'distinguishing aspects that are considered important and by which a subject can be assessed'.
5 Differences in understanding exist regarding whether a criterion denotes something that should be attained and is thus formulated as a goal (normative in nature) or whether a criterion denotes an aspect to be measured and thus is formulated as measurement specification (non-normative in nature).
6 There are two approaches to the definition of an FMU: (i) a defined area of forest land on which forest management activities take place (technical approach). This definition is broadly used for measurement purposes, e.g. by the FAO Forest Resources Assessment. (ii) The administrative unit which decides on and subsequently implements activities in relation to forest management (managerial approach). This definition is more appropriate for certification programmes.
7 The FAO 'Tropical Forest Resource Assessment Project' was started by FAO and UNEP in 1978.
8 FMU = administrative unit.
9 Following ISO definitions, the term 'certification' is used in connection with product or process assessment, the term 'registration' is used in connection with site or company assessment. However, the terms are used differently on different continents.

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Legal Frameworks in Criteria and Indicator Approaches 8

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Currently over 100 countries are involved in developing national-level criteria and indicators (C&I). C&I are also used for certification purposes. The concept of legal framework varies in different C&I approaches, but is usually addressed in relatively broad terms. Legal frameworks together with institutional and economic frameworks provide for not only indicators as such but also the main tools for national, regional and local bodies to respond to developments indicated by other C&I.

1 Introduction

In 1989, the International Tropical Timber Organization (ITTO) established a process to develop C&I for sustainable forest management (SFM). After the United Nations Conference on Environment and Development (UNCED) in 1992, the interest in C&I increased rapidly all over the world.

Some international initiatives share a common objective to develop indicators and to describe, monitor, evaluate and report progress towards SFM at the national level. In many initiatives, countries are urged to develop additional national indicators and, furthermore, to develop indicators at sub-national and forest management unit (FMU) level.

Another use for C&I is certification of forest management. For some customers and non-governmental organizations (NGOs), the credibility of government monitoring of the quality of forest management is low because government is not viewed as being independent and impartial, or because its management standards are not sufficiently enforced (Palmer, 1996).

Certification is a procedure in which an independent third party inspects a product, method or service against certain predetermined requirements, criteria, and, if these criteria are fulfilled, provides a written certificate indicating the conformance.

For producers and suppliers, forest certification is a market-based incentive to improve performance, a marketing tool and a means of satisfying customers' needs and environmental NGOs' (ENGOs) expectations. For customers, certification offers increased credibility. Ecolabelling of the products is one way to achieve some of the objectives.

In certification, the criteria are related to both environmental management systems and performance levels.

Legal frameworks play an important role in the various C&I initiatives. In this chapter, the term 'legal framework' is interpreted broadly, to cover also guidelines, recommendations, etc. In practice, legal frameworks are always complemented by institutional and economic frameworks, which are not dealt with here.

2 International initiatives to develop indicators for monitoring and reporting progress towards SFM at national level

2.1 ITTO criteria for sustainable tropical forest management

The process involved representatives from producer and consumer countries, timber trade as well as intergovernmental and NGOs. A set of five national-level criteria and 27 indicators as well as six FMU-level criteria with 23 indicators was developed in 1991 (ITTO, 1997). An updated set of C&I, including also biological biodiversity and other non-timber values, was published in 1998. It has seven criteria and 56 indicators at national level for natural tropical forests. At the FMU level, 45 indicators are used (Criteria and Indicators . . ., 1998).

2.2 The Helsinki Process

The development of the Pan-European C&I is known as the Helsinki Process. In June 1993, 1 year after UNCED, the Second Ministerial Conference on the Protection of Forests in Europe was held in Helsinki, Finland. The conference was attended by over 200 policy makers and scientists from 37 countries and the European Community, as well as a number of international intergovernmental and NGOs. As a result of the follow-up work of this conference, a pan-European set of six criteria and 27 quantitative indicators for SFM was agreed upon. Each participating country was urged to develop national indicators to supplement the pan-European ones (Costa Leal, 1997).

In January 1995, some 100 examples of supplementary descriptive indicators were introduced, covering the policy instruments (legal/regulatory framework, institutional framework, economic framework and informational means) used for each criterion in order to enhance the sustainable management of forests. Each country may design the descriptive indicators for their own conditions when creating their national C&I (European Criteria and . . ., 1995).

A proposal for Pan-European Operational Level Guidelines was developed in 1998 (Proposal for . . ., 1998).

2.3 The Montreal Process (Santiago Declaration)

In 1993, under the auspices of the Conference on Security and Cooperation in Europe (CSCE), more than 150 scientists and forestry experts from 40 CSCE states, and 13 international NGOs and ENGOs, held a conference in Montreal, Canada. The follow-up work, known as the Montreal Process, currently involves Argentina, Australia, Canada, Chile, China, Japan, the Republic of Korea, Mexico, New Zealand, the Russian Federation, the USA and Uruguay. A consensus has been reached on seven national-level criteria and 67 indicators for SFM for non-European temperate and boreal forests, called the Santiago Declaration (Progress on . . ., 1997).

2.4 The Tarapoto Proposal

The Tarapoto Proposal was developed by the eight signatory countries to the Amazon Cooperation Treaty at a workshop held in Tarapoto, Peru, in 1995. Representatives from Bolivia, Brazil, Peru, Surinam and Venezuela agreed on a set of seven national-level criteria and 47 indicators for Amazonian forests. Furthermore, four criteria and 23 indicators were identified for management unit level as well as one global criterion with seven indicators (Carazo, 1997; Wijewardana *et al*., 1997).

2.5 Dry-Zone Africa

In 1995, the Food and Agriculture Organization of the United Nations (FAO) and the United Nations Environmental Programme (UNEP) hosted a meeting of experts, involving 27 sub-Saharan countries to develop C&I for forests in Dry-Zone Africa. The seven national-level criteria and 47 indicators identified were endorsed for further development by the African Forestry and Wildlife Commission (Taal, 1997; Wijewardana *et al.*, 1997).

2.6 The Near East

For the Near East Region, seven national-level criteria and 65 indicators were developed at an expert meeting hosted by FAO and UNEP in 1996. The results of the meeting were endorsed in principle for further development by the FAO Near East Forestry Commission (El-Lakany, 1997; Wijewardana *et al.*, 1997).

2.7 The Central American Process of Lepaterique

In 1997, FAO and the Central American Commission for Environment and Development held an expert meeting to develop C&I for Central America. The meeting proposed eight criteria and 53 indicators for application at national level, as well as others applicable at the regional level (Blas Zapata, 1997).

2.8 African Timber Organization initiative

In cooperation with the Center for International Forestry Research (CIFOR) and based on field tests in Côte d'Ivoire (1995) and Cameroon (1996), the first draft of African Timber Organization (ATO) C&I was compiled with five principles, two sub-principles, 28 criteria and 60 indicators. Additional testing was planned in Congo, Gabon, Zaire and Ghana. Out of the 187 million ha of African forest cover, 87% is covered by the 13 member states of the ATO and is mainly dense tropical forest (Garba, 1997; Wijewardana *et al.*, 1997).

2.9 CIFOR

CIFOR has field tested C&I in Brazil, Cote d'Ivoire, Indonesia, Germany and USA (southwest Idaho) (CIFOR, 1998).

3 Initiatives to develop indicators for forest certification

3.1 Forest Stewardship Council

The Forest Stewardship Council (FSC) is an international non-profit organization founded in 1993 to support environmentally appropriate, socially beneficial and economically viable management of the world's forests (FSC, 1998a).

FSC has developed ten principles and 52 criteria for forest stewardship (revised in 1996, FSC, 1998b). It also accredits certification organizations. For each country, national standards are developed by national working groups. These standards are used in the actual certification process.

The total area certified is 10 million ha, made up from Sweden 3.2, Poland 1.7 and USA 1.4 million ha (as of June 1998: FSC, 1998c). The FSC product label is internationally recognized.

3.2 International Organization for Standardization

The International Organization for Standardization (ISO) is a worldwide federation of national standards bodies from over 100 countries, one from each country (ISO, 1998a).

ISO 14001, the Environment Management Systems standard, discusses management issues like procedures, organization, competence, etc. Only two performance requirements are defined, namely compliance with legislation and commitment to continuous improvement. Additional performance levels may be set by the organization.

A technical report, based on 2 years' work by the ISO Forestry Working Group, provides a link between the management system approach of ISO 14001 and the range of forest policy and forest management performance objectives, including principles and C&I of SFM that a forestry organization can consider (ISO, 1998b). The report does not add any specific forestry requirements to ISO 14001, or establish performance levels for forest management.

No product label exists for ISO 14001.

3.3 Other standards

In 1996, national SFM System Standards were established in Canada. The Canadian Standards Association (CSA) standards are consistent with the ISO 14001 Environment Management System standard, but also require public participation and performance indicators (The CSA Standards . . ., 1998).

National certification standards are being developed on a multi-stakeholder basis in e.g. Brazil, Finland (Forest Certification in Finland, 1998), Malaysia, Norway (The 'Living Forests' Standards . . ., 1998) and the UK.

The Dutch Keurhout is an association formed by Dutch companies and members of the trade which enjoys government support. Keurhout does not certify forests itself but confirms what certificates satisfy the Keurhout criteria as defined by the Dutch government in 1997: 'Government Paper on Timber Certification and Sustainable Forest Management (minimum requirements)' (Keurhout, 1997; Certification Schemes, 1998).

Country-of-origin labelling has been adopted e.g. in Germany, Austria, Switzerland and the UK (Certification Schemes, 1998).

4 The concept of legal framework in main C&I approaches

4.1 ITTO

In the ITTO approach, legislative aspects are dealt with under Criterion 1: Enabling Conditions for Sustainable Forest Management, which also addresses policy, economic conditions, incentives, research, education, training and mechanisms for consultation and participation. Many of the indicators are descriptive. The indicator related to Policy and Legal Framework contains the following components (Criteria and Indicators . . ., 1998):

1.1 Existence of a framework of laws, policies, and regulations to govern:
- **a.** national objectives for forest including production, conservation and protection,
- **b.** the establishment and security of the permanent forest estate,
- **c.** land tenure and property rights relating to forests,
- **d.** the control of forest management,
- **e.** the control of forest harvesting,
- **f.** the control of encroachment,
- **g.** the health and safety of forest workers, and
- **h.** the participation of local communities.

4.2 Helsinki Process

In the Helsinki Process, no fixed indicators are in use for a legal framework. Each country is urged to address the legal and regulatory framework related to each of the six criteria with the help of provisional descriptive indicators (see Appendix 8.1). Thus, the set of descriptive indicators varies by country.

By definition the legal/regulatory framework in the Helsinki process comprises legal regulations (prohibitions, permissions and obligations) in the form of laws and decrees. They are passed by parliament or councils of state and are binding. Provision of infrastructure by the state is also included.

The other descriptive indicators contain related information in the broader sense of legal framework. In fact even an institutional framework, an economic policy framework and financial instruments as well as informational means are all to a great extent established through legislative and regulatory processes.

4.3 Montreal Process

In the Montreal Process, legal framework is dealt with under a separate criterion. The scope of the term 'legal framework' is wider in comparison to, e.g., the Helsinki process, covering also guidelines, etc. The legal framework-related indicators under criterion 7, 'Legal, institutional and economic framework for forest conservation and sustainable management of forests', are (Montreal Process . . ., 1997):

Extent to which the legal framework (laws, regulations, guidelines) supports the conservation and sustainable management of forests, including the extent to which it:
a. clarifies property rights, provides for appropriate land tenure arrangements, recognizes customary and traditional rights of indigenous people, and provides means of resolving property disputes by due process;
b. provides for periodic forest-related planning, assessment and policy review that recognizes the range of forest values, including coordination with relevant sectors;
c. provides opportunities for public participation in public policy and decision making related to forests and public access to information;
d. encourages best practice codes for forest management;
e. provides for the management of forests to conserve special environmental, cultural, social and/or scientific values.

4.4 Forest Stewardship Council

The six criteria under FSC Principle 1, compliance with laws and FSC principles, are listed in the following (FSC, 1998b):

1. Forest management shall respect all national and local laws and administrative requirements.
2. All applicable and legally prescribed fees, royalties, taxes and other charges shall be paid.
3. In signatory countries, the provisions of all binding international agreements such as CITES, ILO Conventions, International Tropical Timber Agreement (ITTA) and Convention on Biological Diversity, shall be respected.
4. Conflicts between laws, regulations and the FSC Principles and Criteria shall be evaluated for the purposes of certification, on a case by case basis, by the certifiers and the involved or affected parties.
5. Forest management areas should be protected from illegal harvesting, settlement and other unauthorized activities.
6. Forest managers shall demonstrate a long-term commitment to adhere to the FSC Principles and Criteria.

Additional criteria related to legal framework, listed under other principles, include, e.g.:

- Mechanisms to resolve disputes over tenure claims and use rights (criterion 2.3)
- Forest management should meet or exceed all applicable laws and/or regulations covering health and safety of employees and their families (4.2)
- The rights of workers to organize and voluntarily negotiate with their employers shall be guaranteed as outlined in ILO Conventions 87 and 98 (4.3)
- Mechanisms for resolving grievances and for providing fair compensation in the case of loss or damage affecting the legal or customary rights, property, resources or livelihoods of local peoples (4.5)
- Safeguards for protecting rare, threatened and endangered species and their habitats (6.2)
- Written guidelines prepared and implemented to: control erosion, minimize forest damage during harvesting, road construction and all other mechanical disturbances, and protect water resources (6.5)
- Prohibition of certain pesticides (6.6) and genetically modified organisms (6.7)
- Controlled and monitored use of biological control agents (6.7) and exotic species (6.8).

4.5 Other processes

Other processes usually have one criterion containing a legal framework component. The indicators developed for legal framework are usually expressed in broad terms, like ‘National forest policy, legislation and regulations’. In the Dry-Zone Africa proposal, for instance, the indicator is more specific, naming certain key aspects of legislative framework: ‘Existence of a comprehensive legislative and regulatory framework providing for equitable access to resources, alternative forms of conflict resolution and consideration of land occupancy and cultural rights of local populations’.

5 Discussion

5.1 Indicators at national level

Legal framework, in broad sense, is one of the key issues in the entire concept of SFM. The existence already of state-of-the-art forestry legislation, taking into account not only productive but also ecological and socio-economic issues, is a good sign – or indicator – of a government’s commitment to promote SFM.

Closely related subject areas like land-use planning, nature conservation, occupational safety and health, etc., should also have modern and compatible legislation. It is important that binding international agreements such as Conventions on Biological Diversity, Long-Range Transboundary Air Pollution, Climate Change, International Trade in Endangered Species of Wild Fauna and Flora (CITES), and ILO Conventions as well as the ITTA, are incorporated into national legislation.

Legislation is complemented by regulations, codes of practice, guidelines, instructions, recommendations and standards issued at different levels and having different scopes. Efforts should be made to incorporate this network of pieces of legislation into a comprehensive and logical legal framework.

Some of the legislative elements are binding – with or without sanctions – and some voluntary in nature. Enforcement is crucially important, especially in cases where sanctions are considered necessary.

Codes of practice are sets of regulations or guidelines and have been developed in many countries. In some cases they deal with all the aspects of forestry, e.g. the British Columbia Forest Practices Code that contains both legislative and recommendation level components, and the New Zealand Forest Code of Practice that is a collection of guidelines. Specific codes of practice have been promoted for forest harvesting as well as for occupational safety and health, e.g. by bodies like FAO and ILO (Dykstra and Heinrich, 1996a,b; Code of Practice . . ., 1997).

In the Helsinki Process, it is difficult to compare the legal framework, and descriptive indicators in general, between countries as the set of descriptive indicators varies by country. Another feature of the Helsinki Process approach is that the same laws, regulations, recommendations, processes, institutions, etc., often have an effect on several criteria which may lead to repetition of legislative information in the documentation. On the other hand, this type of approach makes it easier to study each component of SFM in a more comprehensive way.

Current indicators for legal frameworks provide a means to monitor national progress towards SFM. In addition, designing of more detailed check-lists on various aspects of legal frameworks would be advantageous. On the other hand, condensation of detailed sustainable development indicators into indices (Hammond *et al.*, 1995) should also be considered.

5.2 Indicators at FMU level

The use of C&I at the FMU level has two major functions: first, monitoring and adjustment of codes of practice, guidelines and prescriptions, and second, possibly forming a basis for certification. Of course, an SFM-minded forest owner may use C&I to monitor and improve his/her operation without applying for a certificate as well.

In all certification schemes, laws, regulations, administrative requirements and international obligations are expected to be respected. In this respect, national indicators of the legal framework are mostly valid and applicable also at the FMU level. Records of inspections and other forms of law enforcement, could, in addition, be used as indicators at the FMU level. FMU-level planning procedures, operating instructions, recommendations, etc., must be in line with legislative elements above them.

5.3 Legal framework as a way to respond

C&I for SFM are, if used properly by governments with a high level of commitment, an excellent tool for evaluating the current forest policy. If unwanted changes are observed, responsible governments and other actors use their power to take action – often through the legal framework. A participatory approach to updating components of the legal and other policy frameworks is highly recommended in order to increase the motivation and commitment of stakeholders.

Commitment and continuous improvement are built-in components in the concept of SFM – just like they are built-in components in standards for environment management systems.

6 Conclusions

1. Legal frameworks have been incorporated into all initiatives on C&I for SFM
2. Legal frameworks can provide useful indicators at a national level
3. Legal frameworks are a means to respond to unwanted changes indicated by other C&I
4. The use of detailed checklists and indices should be studied
5. The indicators related to legal frameworks have not been harmonized
6. More comparative research is needed on legal frameworks in various countries in order to learn from each other.

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Appendix 8.1. Helsinki process criteria and examples of descriptive indicators

(Source: European Criteria . . ., 1995)

Helsinki Process criteria

1. Maintenance and appropriate enhancement of forest resources and their contribution to global carbon cycles.
2. Maintenance of forest ecosystem health and vitality.
3. Maintenance and encouragement of productive functions of forests (wood and non-wood).
4. Maintenance, conservation and appropriate enhancement of biological diversity in forest ecosystems.
5. Maintenance and appropriate enhancement of protective functions in forest management (notably soil and water).
6. Maintenance of other socio-economic functions and conditions.

Descriptive indicators (respective criterion number in parentheses)

Existence of a legal / regulatory framework, and the extent to which it:

- provides an overall policy framework for conservation and sustainable management of forests (1);
- maintains forest resources and prevents forest degradation (1);
- clarifies property rights and provides for appropriate land tenure arrangements (1);
- supports sustainable management while increasing the growing stock of both merchantable and non-merchantable tree species on forest land available for timber production (1);
- clarifies policies for enhancing the use of forest products for energy (1);
- enforces laws and policies related to maintaining forest health and vitality (2);
- encourages forest owners to practice environmentally sound forestry based on a forest management plan or equivalent guidelines (3);
- provides legal instruments to regulate forest management practices for recreation and the harvesting of important non-wood forest products (3);
- clarifies the concept of management, conservation and sustainable development of forest (4);
- provides for national adherence to international legal instruments (4);
- provides for legal instruments to protect representative, rare or vulnerable forest ecosystems (4);

- provides for legal instruments to protect threatened species (4);
- provides for legal instruments to ensure regeneration of managed forests (4);
- provides for legal instruments to regulate or limit forest management practices in forests protected for infrastructure/protection forests (5);
- provides for legal instruments to regulate or limit forest management practices in areas with vulnerable soils (5);
- provides for legal instruments to regulate or limit forest management practices in favour of water conservation or protection of water resources (5);
- provides for legal instruments to ensure development of the forest sector (6);
- recognizes customary and traditional rights of indigenous people, and provides means of resolving access disputes (6);
- provides for legal instruments for securing income levels in the forest sector (6);
- provides for national programmes for research and professional education (6);
- provides opportunities for public access to information (6);
- provides opportunities for public participation in public policy and decision making on forests (6);
- provides for programmes and management guidelines which recognize cultural heritage in relation to forestry (6).

9

Collaborative Action and Technology Transfer as Means of Strengthening the Implementation of National-level Criteria and Indicators

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Interest in the development and use of criteria and indicators (C&I) as essential tools for sustainable forest management has grown rapidly. At present some 150 countries have or are developing and implementing their own set of such tools, at regional, national and/or forest unit levels, within the framework of a number of ongoing processes and/or initiatives. The Food and Agriculture Organization (FAO) is the focal point for this issue among United Nations (UN) agencies, acting as the 'facilitator' of those processes in accordance with the mandate and priorities and in line with the Organization's role as Task Manager for Chapter 11 of Agenda 21 of the United Nations Conference on Environment and Development (UNCED), 'Combating Deforestation'.

However, in order for the implementation of criteria and indicators for sustainable forest management to be successful, all parties involved (UN agencies, ongoing processes, governments and non-government organizations, the private sector including development banks, academic and research institutions) must join efforts to collaborate, transfer and share information, know-how, techniques and methodologies for measuring, assessing and monitoring the indicators with those countries and initiatives that are in need of assistance.

1 Introduction

In follow-up to commitments made at UNCED in 1992, numerous countries have participated in international processes aimed at the definition of criteria for sustainable forest management (SFM) at the regional, national and forest management unit (FMU) levels. Such criteria have proven to be useful tools in country-driven efforts to include and analyse a wide array of forest-derived values within a common conceptual framework. Corresponding indicators are also under development to help quantitatively and qualitatively define these criteria and to allow countries to monitor the effects and analyse trends of management interventions over time, with a view to their progressive improvement to meet defined goals and commitments.

FAO has been involved and instrumental in catalysing and helping pursue a number of international, regional and eco-regional initiatives on C&I for SFM in accordance with its mandate and priorities, and in line with the Organization's role as Task Manager for Chapter 11 of Agenda 21 of UNCED, 'Combating Deforestation'. It is also the focal point for the issue among UN agencies in the work of the Inter-Governmental Panel on Forests (IPF)[1] of the Commission on Sustainable Development.

2 Progress in development and implementation

Commitments made by countries at UNCED to implement the 'Forest Principles' are not legally binding. Nevertheless interest in the development, testing and implementation of C&I for SFM has steadily grown, and currently some 150 countries are developing and implementing their set of C&I within the framework of a number of ongoing dynamic regional and eco-regional initiatives. A summary of the main ongoing international processes is given in Table 9.1.

2.1 Current status of C&I for SFM

Pan-European Process (Box 9.1)

The activities of the Pan-European or Helsinki process were reviewed at the Third Ministerial Conference on the Protection of Forests in Europe held in Lisbon, Portugal (2–4 June 1998). Ministers responsible for forestry adopted, endorsed and encouraged countries to vigorously pursue the implementation of the six national-level criteria developed by member countries and the European Union (EU). Furthermore, they endorsed the 'Pan-European Operational Level Guidelines for Sustainable Forest Management' (Ministry of Agriculture and Forestry, 1996; Third Ministerial Conference, 1998).

Montreal Process (Box 9.2)

The Montreal Process has promoted discussions and consultations within its 12 member countries aimed at refining and expanding information contained in the 'First Approximation Report on Implementation', published by its

Table 9.1. Summary of main ongoing international processes on C&I for SFM.

Process	Number of countries	Number of criteria	Number of indicators	Forest (thousand hectares)	Vegetation coverage
Pan-European	38	6	27	904,577	Boreal and temperate
Montreal	12	7	67	500,000	Boreal and temperate
Tarapoto	8	1/7/4	7/47/22	540,000	Amazonian basin
Dry-Zone Africa	28	7	47	278,021	Sub-Saharan forests
Near East	30	7	65	69,895	Dry-zone forests
C.A. Lepaterique	7	4/80	40/53/0	19,631	All types of forests
ITTO	29	7/7	66/44	1,305,046	Tropical forests
ATO	13	26	60	301,468	African tropical

Figures separated by a slash (/) refer to national-level C&I; the second figure refers to FMU-level. Figures separated by (/ /) include global-, national- and FMU-level C&I, respectively.

The 'forest area' shown in the fifth column should be interpreted only as a general indication of the order of magnitude of the area which could, potentially, be included in SFM activities by countries concerned, using the C&I which have been agreed upon within the international processes in which they participate. More accurately, the area figures might be considered as a demonstration of the challenge countries face to sustainably manage their forests in accordance with present-day concepts.

Forest areas taken from FAO (1993). Total hectare figures do not include forested area for Sao Tome et Principe as it was not available.

Box 9.1.

The 'Pan-European' (or Helsinki) Process focuses on the development of C&I for the sustainable management of forests in 38 European countries. In principle, it includes boreal, temperate and Mediterranean-type forests. The European countries have agreed on six common criteria, 27 quantitative indicators and a number of descriptive indicators for SFM at the regional and national levels.

Member countries: Albania, Austria, Belarus, Belgium, Bosnia-Herzegovina, Bulgaria, Croatia, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Iceland, Ireland, Italy, Latvia, Lithuania, Luxembourg, Malta, Moldavia, Monaco, The Netherlands, Norway, Poland, Portugal, Romania, the Russian Federation, Slovak Republic, Slovenia, Spain, Sweden, Switzerland, Turkey, Ukraine and the UK.

Technical Advisory Committee (Montreal Process, 1997). The report provides an overview of the perceived relevance of the national-level criteria and indicators developed within this process to the conditions, needs and priorities of individual participating countries, and reports on the availability of corresponding data.

Tarapoto Proposal (Box 9.3)

With a view to further develop the Tarapoto Proposal, 'National Consultations for Validation' have been organized by member countries. Through these consultations, each country analysed and systematically evaluated the relevance of the criteria and the applicability of the indicators (at the regional, national and FMU levels) developed within the framework of the Tarapoto Proposal in the light of economic, ecological, social, political and institutional conditions and needs of the Amazon region (Tarapoto Proposal, 1995; and Gobierno de Finlandia/Tratado de Cooperación, 1997).

Near East Process (Box 9.4)

Activity of this Process includes a regional workshop held in Cairo June/July 1997 and attended by the Arab Centre for Studies of Arid Zones and Dry Lands, ACSAD, and the Arab Organization for Agricultural Development, AOAD. During this workshop, national coordinators first reported on progress in the

Box 9.2.

The 'Montreal Process' deals with C&I for SFM in temperate and boreal forests outside Europe. The 12 participating countries have agreed on a set of seven, non-legally binding, criteria and 67 indicators for SFM for national implementation.

Member countries: Argentina, Australia, Canada, Chile, China, Japan, Republic of Korea, Mexico, New Zealand, the Russian Federation, Uruguay and USA.

Box 9.3.

The 'Tarapoto Proposal of C&I for Sustainability of the Amazon Forest' is sponsored by the Amazon Cooperation Treaty. The eight participating countries proposed one criterion and seven indicators of global concern. Furthermore, it identifies seven criteria and 47 indicators for implementation at the national level. For the FMU level, the process recognizes four criteria and 22 indicators.

Member countries: Bolivia, Brazil, Colombia, Ecuador, Guyana, Peru, Suriname and Venezuela.

analysis of applicability and testing or implementation of the national-level C&I developed within the framework of the Near East Process, and secondly, presented a consolidated proposal for coordinated future action, taking into consideration the results of a country-based survey of the availability, periodicity and reliability of data corresponding to the proposed common C&I and the capacity of countries to undertake the work required (FAO/United Nations Environmental Programme (UNEP), 1996; FAO 1997).

Dry-Zone African Process (Box 9.5)

Subsequent to the FAO/UNEP Expert Consultation in 1995 (21–24 November), UNEP and FAO assisted the 28 countries participating in the Dry-Zone Africa Process in the organization of a follow-up workshop in Nairobi (November 1997). Regional organizations, including the Permanent Interstate Committee for Drought Control in the Sahel, CILSS; the Inter-Governmental Authority on Drought and Development, IGADD; and the Southern African Development Community, SADC collaborated in the workshop. Countries presented an analysis of the applicability of the proposed national-level C&I, actual or potential availability of data and national capacities for collection and analysis of data. A consolidated Plan of Action was prepared and agreed upon by countries present (UNEP/FAO, 1995, 1997).

Box 9.4.

In the FAO/UNEP Expert Meeting on C&I for SFM for the Near East (Cairo, Egypt 15–17 October 1996), countries identified seven criteria and 65 indicators for SFM at the regional and national levels.

Member countries: Afghanistan, Algeria, Baharain, Cyprus, Djibouti, Egypt, Islamic Republic of Iran, Iraq, Jordan, Kuwait, Kyrgyz Republic, Lebanon, Libya, Malta, Mauritania, Morocco, Oman, Pakistan, Qatar, Kingdom of Saudi Arabia, Somalia, Sudan, Syria, Tadjikistan, Tunisia, Turkey, Turkmenistan, United Arab Emirates and Yemen.

Box 9.5.

The UNEP/FAO Expert Meeting on C&I for SFM in Dry-Zone Africa (Nairobi, Kenya, 21–24 November 1995) identified seven criteria and 47 indicators for SFM at the national level.

Member countries: Dry-Zone Africa Process: 28 countries. CILSS (nine countries): Burkina Faso, Cape Verde, Chad, Gambia, Guinea Bissau, Mali, Mauritania, Niger and Senegal. IGADD (seven countries): Djibouti, Eritrea, Ethiopia, Kenya, Somalia, Sudan, Uganda. SADC (12 countries): Angola, Botswana, Lesotho, Malawi, Mauritius, Mozambique, Namibia, South Africa, Swaziland, Tanzania, Zambia, Zimbabwe.

Lepaterique Process of Central America (Box 9.6)

The Central American Commission on Environment and Development (CCAD), through its Technical Secretariat, the Central American Council on Forests and Protected Areas (CCAB-AP), continues to review and test regional-, national- and FMU-level C&I proposed by the seven-member countries (CCAD/FAO/CCAB-AP, 1997). The regional Expert Meeting, organized within the framework of an FAO project in January 1997, was followed by two sub-regional workshops and seven national seminars supported by FAO and the CCAD. FAO assisted the CCAB-AP in its search for donors to fund the implementation of FMU-level C&I in countries in the region. FAO is also helping countries in the Caribbean to join the initiative, through funds likely to become available from the EU.

International Tropical Timber Organization (ITTO) (Box 9.7)

ITTO is a major contributor and an important participant in sustainable tropical forest management. The assessment of forest management status and the transfer of environmentally sound technologies, along with capacity building, rank high in ITTO's agenda. Through an Expert Panel established in 1997 by the ITTC, the 1992 'ITTO Criteria for Sustainable Tropical Forest

Box 9.6.

An FAO/CCAD Expert Meeting on C&I for SFM in Central America (Tegucigalpa, Honduras, 20–24 January 1997) and officially known as the 'Lepaterique Process of Central America' identified four criteria and 40 indicators at the regional level and eight criteria and 53 indicators at the national level. This regional Expert Meeting was later on complemented by two FAO/CCAD supported sub-regional meetings and seven national seminars on country-level implementation and on the identification of C&I at the FMU level.

Member countries: Belize, Costa Rica, El Salvador, Guatemala, Honduras, Nicaragua and Panama.

Box 9.7.

In the FAO/ITTO Expert Meeting (Rome, 13–16 February 1995) the possibilities of harmonizing ongoing initiatives related to C&I for SFM were analysed. The meeting agreed on the need to widely exchange information, know-how and experience between ongoing initiatives in order to ensure comparability between them and to avoid needless duplication of effort. The meeting also stressed the need to allow current initiatives to pursue their aims unimpeded, reflecting the different national and/or regional environmental and socio-economic conditions.

ITTO Member countries: 55: 29 producers, 25 consumers and the EU.

Management' was revised, taking into account recent trends and international developments in the field (ITTO, 1998, 2000).

A 'Manual for the Application of Criteria and Indicators for Sustainable Management of Natural Tropical Forests' was issued in 2000.

To November 2000, ITTO had sponsored 577 fellowships for training in relation to sustainable tropical forest management, to nationals of over 30 countries. Direct financial support to related projects has also amounted to millions of dollars over the years. In 1993 this assistance was $15.5 million (IPF/Interagency Task Force on Forests (ITFF), 1997).

3 Opportunities for development in the light of capacity building

'Technology transfer' related to C&I in support of SFM is referred to in this document as seen by the Inter-Governmental Forum on Forests (IFF); it encompasses techniques as well as methods, technical knowledge and information sharing. Consequently, it is a component of a wide range of programmes and projects at various levels from research and scientific information to technical cooperation and extension.

3.1 Collaborative action for developing and strengthening the implementation of national-level C&I

International collaboration

FAO, in its capacity of UN focal point for action in the forestry sector, continues to closely follow all ongoing international processes, support efforts by countries and regions in response to requests received, and helps disseminate information and know-how among the processes. These activities are partially facilitated as they are carried out through 'National Coordinators' who have been identified for each country and/or region, for example for the Near East, Dry-Zone Africa and Lepaterique Processes.

Also in this regard, the XI World Forestry Congress organized by the Government of Turkey in collaboration with FAO in Antalya in October 1997 placed major emphasis on present-day concepts of SFM. The overall programme of the congress was arranged in accordance with the sustainability criteria commonly identified in the international processes. One of the main plenary sessions dealt with this issue, including presentations by representatives of all major ongoing processes.

The activities of the ad hoc ITFF, which supports the work of the IPF and helps to strengthen and streamline concerted action by international agencies, will continue in support of the work of the IFF. The first meeting of the ITFF following the establishment of the IFF was recently held in August 1998 in Geneva, Switzerland.

Other UN agencies involved in technology transfer and capacity building (Table 9.2)

One of the results of the 4th Session of the IPF (June l997) was a proposed plan of action, drafted by the ITFF Panel. The proposed 'Plan – Forest 21' has as its ultimate goal implementation of IPF's proposal for action through well

Table 9.2. Some UN agencies and convention secretariats promoting, supporting and assisting countries in the development and implementation of C&I for SFM.

Programme element	Facilitator	FAO	UNDP	UNEP
I.1: Progress through national forests and land-use programmes	FAO		X	X[a]
I.2: Underlying causes of deforestation and forest degradation	UNEP	X	X	
I.3: Traditional forest-related knowledge[b]	CBD	X	X	X
I.3a: Programme of work on forest biological diversity	CBD	X	X	X
I.4.1: Fragile ecosystems affected by desertification and drought	FAO		–	X
I.4.2: Impact of air-borne pollution of forests	FAO		–	X
I.5: Needs and requirements of countries with low forest cover	UNEP	X	–	
II: International cooperation in financial assistance and technology transfer for SFM	UNDP	X		X
III.1(a)/1: Assessment of the multiple benefits of all types of forests	FAO		–	X
III.1(b)/2: Forestry research	(CIFOR)	–	–	–
III.1(b): Methodologies for the proper valuation of the multiple benefits of forests	WBG	X	–	X
III.2: Criteria and indicators for SFM	FAO		X	X
IV. Trade and environment relating to forest goods and services	(ITTO)	X	–	X

FAO, Food and Agriculture Organization of the United Nations; UNDP, UN Development Programme; UNEP, UN Environment Programme; UNESCO, UN Educational, Scientific and Cultural Organization; WBG, World Bank Group; UNIDO, UN Industrial Development Organization; DPCSD, UN Department for Policy Coordination and Sustainable Development; UNSO, UN Special Office to Combat Desertification and Drought; UNCTAD, UN Conference on Trade and Development; UN/ECE, UN Economic Commission for Europe; CCD, Convention executed and coordinated activities by ITFF members in association with other international organizations, multilateral agencies and instruments. The plan, which addresses needs at the regional and international levels, has as one of its aims to 'recognize opportunities for participation by other potential partners such as governments, non-governmental organizations (NGOs) and other international organizations, for a more effective response to IPF's proposal for action'.

Three of the more relevant 'Programme Elements' of the plan which specifically refer to C&I in the framework of technology transfer and capacity building are (IPF/ITFF, 1997):

Some of the major United Nations partners									
UNESCO	WBG	UNIDO	DPCSD	UNSO	UNCTAD	UN/ECE	CCD	CBD	CCC
–	X	X	–	–	–	–	X	X	–
–	–	–	–	–	–	–	X	X	–
X	X	X	X	–	X	–	X		X
X	–	–	–	–	–	–	X		X
–	–	–	–	X	–	–	X	X	X
–	–	–	–	–	–	–	–	X	X
–	X	–	–	–	–	–	X	X	–
–	X	–	–	–	–	–	–	–	–
–	–	–	–	–	–	X	–	–	–
–	–	–	–	–	–	–	–	–	–
–		–	–	–	–	–	–	X	–
X	X	–	–	–	–	–	–	X	–
–	–	–	–	–	X	–	–	–	–

on Combating Desertification; CBD, Convention on Biological Diversity and CCC, Convention on Climate Change; ILO, International Labour Organization; WIPO, World Intellectual Property Organization; WTO, World Trade Organization.

[a]An 'X' means that the agency or secretariat explicitly appears as a 'major' partner for the programme element; a '–' means it does not fully appear as one although it may eventually become a 'partner'. Names in (): Non-UN facilitators.

[b]Includes also ILO, WIPO and WTO.

1. *Programme Element I.4:* 'Part Two: Impact of Air-borne Pollution on Forests'. The objective of this element is to reduce air-borne pollution and its effect on forests through the promotion of international cooperation in the exchange of information and experience on assessment and monitoring of the corresponding effects of pollution in forests. This element hopes to achieve its goal through the development of methods to measure such effects in the context of national-level C&I for SFM. FAO is the facilitator for this element; major partners are the UNEP, the Convention on Climate Change (CCC), the Convention on Biological Diversity (CBD) and the Convention to Combat Desertification (CCD) through the United Nations Special Office to Combat Desertification and Drought (UNSO).
2. *Programme Element II:* 'International Cooperation in Financial Assistance and Technological Transfer for Sustainable Forest Management'. The overall objective is to establish successful models which may improve the effectiveness of existing international cooperation, partnership and financial resources. Innovative mechanisms are being sought to finance SFM, and especially the transfer of environmentally sound technology, mainly through improved capacity building. The news from this initiative has been encouraging since 1991. The facilitator for this element is the United Nations Development Programme (UNDP), with UN partners UNEP and FAO.
3. *Programme Element III.2:* 'Criteria and Indicators for Sustainable Forest Management'. The overall short- and long-term objectives of this element are to 'visibly and convincingly move towards' SFM by the year 2000 in line with and complementing ITTO's Objective 2000 and 'to promote attainment of sustainable forest management in all countries and in all kinds of forests', respectively. The element calls for providing countries with the technical and financial means to better develop, test and implement C&I in order that the countries may reach a common international understanding that these tools are indeed important in the search for sustainable forest management. FAO is the facilitator for this element and major partners include UNEP, UNDP, the United Nations Educational, Scientific and Cultural Organization (UNESCO) and the CBD.

A fourth 'Programme Element' that indirectly deals with the issue of how, when and which C&I for SFM countries can measure and report on in relation to their capacity, experience and 'know-how' is the following:

Programme Element III.1(a): 'Part One: Assessment of the Multiple Benefits of All Types of Forests'. One of the main issues of this element is the assessment and periodic evaluation of global forest resources (including information on the goods and services that forests provide) which may provide information necessary for planning and evaluating effects (quantitatively and qualitatively) of various forestry activities. This element addresses the need for better coordination and prioritization of forestry data collection as part of national forest inventories and the avoidance of overlap (in data collection and analysis) between forest and other related information systems considering countries' limited financial and technical capacities to collect it. FAO has the responsibility for this element through its Forest Resources Assessment Programme, FRA 2000.

Other UN bodies such as the United Nations Industrial Development Organization (UNIDO) and the United Nations Statistical Division (UNSD) also promote SFM by undertaking surveys, and collecting, analysing and disseminating data relevant to such management.

Other parties involved

Examples of other institutions that are directly or indirectly involved in technology transfer and improving countries' capacity to develop and apply C&I for SFM at the national and/or field levels, are given in Table 9.3.

THE INTERNATIONAL UNION OF FORESTRY RESEARCH ORGANIZATIONS (IUFRO) It was particularly encouraging to see IUFRO, through its Inter-Divisional Task Force on Sustainable Forest Management, organize a major conference on indicators for SFM in collaboration with the Center for International Forestry Research (CIFOR) and FAO in 1998. It was an excellent example of collaboration between agencies to improve communication and exchange of information among those involved in C&I.

IUFRO is a major partner, along with other international research institutions, mentioned in Part 2: Forestry Research of Programme Element III.1(a) as a complement of this element's Part One: 'Assessment of the Multiple Benefits of All Types of Forests'. IPF reports highlight the need for more information and knowledge of SFM. The same reports emphasize the need for more international coordination in research and for 'new cultures' of research, putting emphasis on demand-driven, holistic and integrated approaches. This was also one of the recommendations of the Kochi Workshop on Integrated Application of Sustainable Forest Management Practices (Kochi, Japan; 22–25 November 1996). IUFRO's role in the 1998 conference was very appropriate, as one of the overall objectives of the conference was to identify gaps in knowledge and priorities for research to underpin action in the field of SFM.

CIFOR Participation of private and other non-government sectors appears to be even more important for the implementation of C&I at the FMU level. For example, CIFOR has undertaken a multi-year project financed by the EU, other donors and previously by the Deutsche Gesellschaft für Technische Zusammenarbeit (GTZ). The project focused on the identification and testing of C&I for SFM at the field level. CIFOR expects that results of this research project, carried on in Brazil, Cameroon, Côte d'Ivoire and Indonesia, will assist forest managers in the development of methodologies and scientifically sound,

Table 9.3. Other non-UN institutions supporting and assisting countries in the development and implementation of C&I for SFM.

	Other major partners					
Programme element	ITTO	IUFRO	CIFOR	RDB	BD	EFI
I.1: Progress through national forests and land-use programmes	–	–	–	X	–	–
I.2: Underlying causes of deforestation and forest degradation	–	–	–	–	X	–
I.3: Traditional forest-related knowledge	X	–	–	X	–	–
I.3a: Programme of work on forest biological diversity	–	–	–	–	–	–
I.4.1: Fragile ecosystems affected by desertification and drought	–	–	–	–	–	–
I.4.2: Impact of air-borne pollution of forests	–	–	–	–	–	–
I.5: Needs and requirements of countries with low forest cover	–	–	–	X	X	–
II: International cooperation in financial assistance and technical transfer for SFM	X	–	–	X	X	–
III.1(a)/1: Assessment of the multiple benefits of all types of forests	X	X	X	–	–	–
III.1(a)/2: Forest research	–	X	Fa	–	–	X
III.1(b): Methodologies for the proper valuation of the multiple benefits of forests	–	–	X	–	X	X
III.2: Criteria and indicators for SFM	X	X	X	X	–	–
IV: Trade and environment relating to forest goods and services	Fa	–	–	–	–	–

RBD, Regional Development Banks; BD, Bilateral Donors; EFI, European Forestry Institute; 'Fa', Facilitator.

I.5: Includes also ODA-UK, Overseas Development Agency, UK; now DFID, Department for International Development.

III.1(a)/2: Includes also ICRAF, International Centre for Research in Agroforestry; IBFRA, International Boreal Forest Research Association; and WCMC, World Conservation Monitoring Center.

III.1(b): Includes also HIID, Harvard Institute for International Development.

III.2: Includes also the IUCN, The World Conservation Union and WWF, World Wide Fund for Nature.

practical guidelines for the assessment and monitoring of SFM in the field. Early efforts of the project concentrated on criteria and related indicators dealing with biological diversity, with socio-economic and (to a lesser degree) with economic aspects. Subsequently attention expanded to identify policy environments which favour and promote SFM, and to 'adaptive co-management systems', through which the rights and responsibilities of all stakeholders are identified, acknowledged and incorporated into overall forest management planning and implementation.

To complement CIFOR's project, the Australian Centre for International Agricultural Research (ACIAR) and CIFOR, in cooperation with the Kerala Forest Research Institute, India, commenced testing C&I for the sustainability of tropical forest plantation management in Kerala, India.

THE AFRICAN TIMBER ORGANIZATION (ATO) (Box 9.8) This organization has been active over the past few years in identifying relevant 'principles' and C&I for SFM through testing at the FMU level, with the main aim of developing appropriate tools for classifying, qualifying and certifying forest management in its member nations (ATO, 1998).

CUBA At its own initiative, and partially assisted by the CCAB-AP of the Central American Lepaterique Process, this country has already developed its own set of national-level C&I and is now in the process of implementation.

SUPPORTERS Examples of high-level support for SFM include the Denver Summit of the G8 countries (Canada, France, Germany, Italy, Japan, Russia, UK and USA) in June 1997, where it was proposed to establish a practical Action Programme on Forests that includes 'implementing national programmes and building capacities for sustainable forest management; establishing networks of protected areas; assessing the state of each nation's forests using agreed upon criteria and indicators; promoting private sector management of forests; and eliminating illegal logging'. The G8 countries met again at the Summit in Birmingham, UK (15–17 May 1998) and agreed on five categories of actions, as part of an Action Programme on Forests, one of which is to continue work with partner countries to build national capacity in order to 'improve scientific underpinning of the economic, social and environmental indicators of sustainable forest management'.

Several bilateral donors, development banks and NGOs also play an important role in technology transfer and capacity building on issues dealing

Box 9.8.

The ATO's main priority since 1994 has been to 'promote the implementation of sustainable forest management in ATO member countries', and 'in accordance with recommendations made at the international level, specially by the Inter-Governmental Panel on Forests' it has chosen to use for its work five principles, two sub-principles, 26 criteria and 60 indicators at the regional and national levels.

Member countries: Angola, Cameroon, Central African Republic, Congo, Côte d'Ivoire, Democratic Republic of Congo, Equatorial Guinea, Gabon, Ghana, Liberia, Nigeria, Sao Tome et Principe and Tanzania.

with sustainable forest management including criteria and indicators (Table 9.3). For example, the present level of investment in forestry for the Overseas Development Agency (ODA) (now Department for International Development, DFID), UK, is 27.2% and the role of the private sector in forestry financing has increased by 60% since 1991. Other regional bodies such as ACSAD and AOAD in the Near East and IGADD, CILSS and SADC[2] in Africa are collaborating with the Near East and the Dry-Zone Africa Processes, respectively.

3.2 Dealing with regional and national capacities to implement C&I

Definitions, terminology and data collection

Following the request by the IPF for FAO to act as 'facilitator' and later stressed by its member nations, efforts have been intensified to assist countries and national institutes to conceptualize, develop and promote the implementation of C&I for SFM at national and eco-regional levels; to help streamline concepts and terminology and thus to ensure compatibility of ongoing and planned international efforts in this field; and to assist countries and regions in monitoring the effects of forest management on forest ecosystems and in improving methodologies to ensure their sustainability; special attention is given to sites marginal to plant growth and fragile ecosystems.

In order to report advances internationally in SFM involving C&I, it is necessary for countries and/or processes to use terminology and concepts that mean the same universally. In this respect the Harmonization Meeting (FAO/ITTO, 1995) reviewed possibilities of harmonizing ongoing initiatives. Besides agreeing on the need for exchange of information, know-how and experience between ongoing initiatives to ensure comparability between them and to avoid wasteful duplication of effort, the meeting also emphasized the need to allow ongoing initiatives to pursue their aims unimpeded, reflecting the different environmental and socio-economic conditions from which they have sprung. This conclusion was also reached at the Intergovernmental Seminar on Criteria and Indicators for Sustainable Forest Management (Ministry of Agriculture and Forestry, Finland, 1996), organized by the Government of Finland in collaboration with FAO in support of the work of the IPF.

In an effort to make collection of field data which quantify indicators more meaningful and efficient, FAO supports and encourages countries to revise the suggested set of national-level C&I. This should lead to a national exercise to prioritize and decide which C&I are or are not relevant to the country, based on its current environmental, ecological, social and economic conditions. It is important that the country has the capacity to measure the selected indicators at the relevant time. Given that most countries, especially the developing ones, have problems in measuring some of the indicators, FAO promotes the collection and reporting of field data of at least those parameters which form part of national forest inventories and whose results countries voluntarily contribute to FAO's Forest Resources Assessment Programme (FRA 2000).

The 'Expert Consultation on Global Forest Resources Assessment 2000' held in Kotka, Finland (June 1996) recognized that the parameters to be included in the Global FRA should be (i) relevant and useful at the international level; and (ii) possible to assess with the available data acquisition tools at acceptable cost. Considering those two main characteristics of 'good parameters', experts identified 15 indicators for SFM which could be assessed through the Global FRA 2000 (FAO, 1996).

The indicators to be looked at in priority refer to the following elements of criteria: forest resources, biological diversity, forest health and vitality, production of wood and forest products, soil and water protection, and social and economic functions. The periodicity of measurement of indicators is another consideration which will depend on countries' technological and maybe even more so on financial conditions. The time span between measurements will reflect the periodic rate of change of the parameter.

Furthermore, the success of a process and/or initiative in the implementation of national-level C&I for SFM depends on two additional main conditions: (i) political endorsement principally from the respective forestry authorities, as is the case of the Pan-European and the Central American Lepaterique Processes and the Tarapoto Proposal; and (ii) processes that preferably operate within the framework of ongoing national forestry projects.

Future initiatives

Given the interest of the Caribbean countries to get involved in the development and implementation of national-level C&I for SFM, FAO submitted a project proposal to the EU covering 15 countries of the region signatories of the Lome Convention.[3]

4 Issues and opportunities for research

4.1 Linkages between national- and FMU-level C&I

The overall aim of the development of both national- and FMU-level C&I is to achieve better forest management over time. Thus, FMU-level C&I should be linked to the national level, and the two levels must be mutually compatible. However, C&I developed at these levels differ in concept and substance. The national-level indicators will contribute towards the development and regular updating of policy instruments (laws, policies, regulations), while trends in the indicators at the FMU-level will help adjust forest management prescriptions over time to meet established national goals (Castañeda, 1997a,b).

4.2 Research needs for the transfer of technology and capacity building

The implementation of C&I is a process that requires constant and repeated testing according to the ecological, environmental, social and economic conditions of countries. Thus research is an important step towards the implementation of these tools, especially at the FMU level. Research must be applied, selective and oriented towards studying mainly those issues that will assess sustainability of forests managed by local communities within their limitations; for example those C&I for SFM dealing with social, economic and policy issues.

Continuing effort should also be made to assist and promote research to develop monitoring systems by which countries may assess trends in forest management and conditions at both national and FMU levels. In this respect, the role that institutions such as CIFOR and the World Bank, as facilitators of Programme Elements III.1(a), Part Two: 'Research' and III.1(b): 'Methodologies for the proper valuation of multiple benefits of forests' respectively, can be decisive.

Finally, it is time to promote inclusion of issues of C&I in academic forestry curricula, mainly universities. Successful implementation of these tools requires a solid basis; success can partially be attained through new foresters who know about and understand the need for C&I as a means of measuring sustainability of forest management in the context of sustainable development. The inclusion of C&I in academic curricula would also enable those universities and other forestry schools to collaborate with institutions already conducting relevant research.

5 Conclusions and recommendations

- Countries not yet actively participating in any of the international processes should be encouraged and assisted to join such activities. In order to avoid duplication of effort and wasteful overlaps, it is proposed that, as a first step, countries be encouraged to review the C&I developed by ongoing processes which are considered by them to most closely correspond to their environmental, social, economic and institutional conditions, with a view to adapting and testing these to meet prevailing regional and national needs and priorities.
- There is a need to ensure continuing dialogue among ongoing and new and emerging processes over the coming years, that C&I developed and implemented by countries within the framework of these processes are mutually compatible and that they contribute towards a common understanding of issues at stake. There is also an urgent need to arrive at common concepts and terminology, and to draw attention to a

continuing, increasing need for international dialogue to facilitate common understanding and compatibility of action between countries and regions.

- As suggested in a meeting of the Montreal Process during the XI World Forestry Congress (Turkey, October 1997) countries which have advanced in respect to the above should share their experiences and assist others towards implementation. Similarly the Pan-European Process is also encouraged to continue providing support, advice and assistance to the more recent processes as requested.
- Countries concerned need to ensure the compatibility of C&I implemented at the national level and those being developed at the FMU level.
- Within the limits of available resources, FAO will continue to ensure the flow and dissemination of information, know-how and technologies among the international processes, thus facilitating comparability and compatibility among them as requested by the IPF. This commitment includes providing support to the follow-up of resolutions such as the ones passed, for example, in the Ministerial Conferences on the Protection of Forests (Lisbon, 2–4 June 1998). The Organization is pleased to draw on the expertise of ongoing processes in order to promote the development and implementation of C&I around the globe. For example, the Pan-European Process helped start and catalyse similar C&I processes in Dry-Zone Africa, the Near East and Central America.
- The level of investment in SFM by both national governments and the private sector is not likely to increase. Thus it is necessary for these key investors to substantially improve their strategies to implement SFM, including taking steps towards establishment of strong partnerships between government institutions, the private sector, bilateral and multilateral development agencies, research institutions, local governments and NGOs (IPF/ITFF, 1997).

Notes

1 The mandate of IPF expired in March 1997. At the fourth and final session of the Inter-Governmental Forum on Forests (IFF), a follow-up mechanism, held in New York in February 2000, the UN Forum on Forests (UNFF, http://www.un.org/esa/sustdev/forests.htm) was established as a new non-legally binding body to facilitate and promote the implementation of proposals for action emanating from IPF and the IFF. Its role will also extend to strengthening the partnership of existing UN and other organizations in implementing sustainable forest management practices worldwide.

2 IGADD: Djibouti, Ethiopia, Kenya, Somalia, Sudan and Uganda; CILSS: Burkina Faso, Cape Verde, Guinea Bissau, The Gambia, Mali, Mauritania, Niger, Senegal and Chad; SADC: Angola, Botswana, Lesotho, Malawi, Mozambique, Namibia, South Africa, Swaziland, Tanzania, Zambia and Zimbabwe.

3 The Caribbean countries being considered under this ‘possible new initiative’ include Antigua and Barbuda, Bahamas, Barbados, Dominica, Dominican Republic, Grenada, Guyana, Haiti, Jamaica, Saint Kitts and Nevis, Saint Lucia, Saint Vincent and Grenadines, and Trinidad and Tobago. Belize and Suriname are excluded as they already form part of the Central American Lepaterique Process and the Tarapoto Proposal, respectively.

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10 Inventory and Forecasting Productive Capacity for Natural Forests

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The productive capacity of natural forests can be monitored against three common indicators: area of forest available for timber production; extent to which harvested areas are regenerated; and level of harvesting for wood and non-wood products compared to the sustainable level. These indicators have been derived from an analysis of five internationally representative approaches to sustainable forest management (SFM).

Although our knowledge of the productive capacity of forests is long established and considerable, there are many opportunities to enhance this knowledge base. This chapter considers these indicators in the context of natural forest inventory and forecasting, and recommends enhancements of and extensions to the existing productive capacity knowledge base.

1 Introduction

This chapter focuses on the inventory and forecasting elements of natural forest management. It considers the research and development needs of the three common indicators of productive capacity, and assumes the availability of a dataset similar to that described in a case study for Victoria, Australia (Department of Natural Resources and Environment, DNRE, 1997a).

(eds R.J. Raison, A.G. Brown and D.W. Flinn)

1.1 Definition of natural forest

Consideration of the appropriate sustainability indicators to describe the productive capacity of natural forests is based on an understanding of what constitutes a natural forest, and the identification of appropriate indicators related to the productive capacity of these forests.

Forests are considered to be 'land with tree cover (or equivalent stocking level) of more than 10% and an area of more than 0.5 ha. The trees should be able to reach a minimum height of 5 m at maturity *in situ*. The forest may consist *either* of closed formations where trees of various storeys and undergrowth cover a high proportion of the ground; *or* of open formations with a continuous vegetation cover in which tree crown cover exceeds 10 percent'. (United Nations – Economic Commission for Europe/Food and Agriculture Organization of the United Nations, UN-ECE/FAO, 1997). Although this definition does not consider forest areas that for various reasons are currently 'non-forested', Prüller (1997) considers that most countries use this definitional approach and describe their closed or open forest formations in terms of similar tree crown cover and mature height.

The definition of a natural forest has been widely debated. Many countries regard their native or locally indigenous forest communities as their natural forests. Simplistically 'natural forests are a subset of forests composed of tree species known to be indigenous to the area' (FAO, 1995). Other countries consider that natural forests should have developed in the absence of human influence (and terms such as primal, primitive and old-growth forest are used to describe them), and that forests of indigenous or other tree species generated after the original forest cover has been disturbed should be classified as secondary forest. Prüller's (1997) comparative terminology study concludes that the use of many synonymous terms causes international confusion, and any definition must clearly identify that 'natural forest is composed of indigenous species, and that its establishment, regeneration and development are done naturally'. He reports that some countries (for example Vietnam) still classify as natural forest areas where some human intervention has occurred, and where forests are enriched by natural or man-made regeneration. In this chapter, the adoption of Prüller's refined definition constrains the discussion of appropriate sustainability indicators for productive capacity of natural forests.

Prüller (1997) suggests that the concepts of natural and plantation forests can be seen as antonyms. Hence approaching the natural forest definitional problem from an alternative perspective may be useful. Plantations are 'forest stands established by planting or/and seeding in the process of afforestation or reforestation. They are either:

- of introduced species (all planted stands), or
- intensively managed stands of indigenous species, which meet all the following criteria: one or two species at plantation, even age class, regular spacing' (UN-ECE/FAO, 1997).

This chapter proposes that the broad 1995 FAO definition is modified as follows:

> Natural forests are a subset of forests (which includes secondary forests, but excludes plantations) composed of tree species known to be indigenous to the area.

1.2 Productive capacity – common sustainability indicators

There are many approaches to the expression of sustainable forest management (SFM). Each major global forest/vegetation zone has a set of criteria and indicators (C&I) considered appropriate for those forests and their management. A comparison of the most common indicators of productive capacity used for national- and regional-level reporting in a number of zones is shown in Table 10.1. The approaches considered were African Timber Organization, International Tropical Timber Organization, the Helsinki and Montreal Processes, and the Tarapoto Proposal.

From the five approaches considered in the table, the three indicators most commonly used for the productive capacity of natural forests were:

- area of forest available for timber production;
- extent to which harvested areas are regenerated; and
- level of harvesting for wood and non-wood products compared to the sustainable level.

Measuring and monitoring these three common indicators can reveal the extent to which current forest management systems are maintaining the productive capacity of natural forests. This chapter describes the Victorian datasets as a case study, and considers the context of each indicator in natural forests. The first two common indicators will be considered in the context of inventory, the third in the context of modelling the forest in order to forecast sustainable levels of forest production.

1.3 Case study – Victoria's productive capacity datasets and modelling systems

The Victorian forest datasets have been selected for this case study because the inventory data are collected in a contemporary framework, recorded in digital format and linked to the forest management planning process by a sophisticated modelling system.

Much of the data needed to describe the productive capacity of natural forests is currently collected as part of everyday forest management. In Australia, natural forest resource inventories tend to focus on the sub-national (State), regional or forest management unit (FMU) levels. Victoria's Statewide Forest Resource Inventory (SFRI) is an example of a multipurpose statewide strategic inventory of natural forests where stand mapping, biodiversity and resource

Table 10.1. Sustainability indicators of productive capacity for a range of regional processes.[a]

ATO	Helsinki Process	ITTO	Montreal Process	Tarapoto Proposal
1.1.1. There exists a map showing the boundaries of the permanent forest estate. 2.1.1. There is a management plan comprising: • definition of the forest area subjected to sustainable forest management.	1.1. Area of forest and other wooded land and changes in area (classified . . . according to . . . ownership structure, age structure and origin of forest).	2.1. Extent (area) and percentage of total land area under: • natural forest • plantation forest • permanent forest estate, and • comprehensive integrated land-use plans. 2.4. Area of the permanent forest estate converted to permanent non-forest use.	2.1.a. Area of forest land and net area of forest land available for timber production.	3a. Extent and proportion of forest lands and forests dedicated to sustainable production in relation to the total permanent production area.
	1.2. Changes in: • total volume of the growing stock • mean volume of the growing stock on forest land • age structure or appropriate diameter distribution classes.	4.1. Extent and percentage of forest for which inventory and survey procedures have been used to define: • the quantity of the main forest products, and • resource rights and ownership.	2.1.b. Total growing stock of both merchantable and non-merchantable tree species on forest land available for timber production.	
2A.2.3. Calculations of allowable cut and rotation period are clearly detailed in the management plan and are consistent with silvicultural standards, increment data, prior inventory and harvestable areas, and are established at levels considered compatible with sustainable production of the forest.	3.1. Balance between growth and removals of wood over the past 10 years.	4.2. Estimate of level of sustainable harvest for each main wood and non-wood forest product for each forest type. 4.3. Quantity (volume) of wood and important non-wood products harvested for each forest type.	2.1.d. Annual removal of wood products compared to the volume determined to be sustainable.	3b. Quantity and proportion of sustainable forest production in comparison with the national total forest production. 3c. Quantity and proportion of units of sustainable production, by area class, in comparison with the national total number of units.

2B.1. Non-timber forest products and their uses are identified. 2B.2. Guidelines for rational harvesting of non-timber forest products are defined and put into practice.	3.3. Total amount of and changes in the value and/or quantity of non-wood forest products (e.g. hunting and game, cork, berries, mushrooms, etc.).	4.2. Estimate of level of sustainable harvest for each main wood and non-wood forest product for each forest type. 4.3. Quantity (volume) of wood and important non-wood products harvested for each forest type.	2.1.e. Annual removal of non-timber forest products (e.g. furbearers, berries, mushrooms, game), compared to the level determined to be sustainable.	3b. Quantity and proportion of sustainable forest production in comparison with the national total forest production. 3c. Quantity and proportion of units of sustainable production, by area class, in comparison with the national total number of units.
2A.3.1. Reforestation is implemented with chosen species in conformity with the specifications of the management plan. 3.1.3. Actions are taken to assure natural regeneration when necessary.	4.5. In relation to total area regenerated, proportions of annual area of natural regeneration.	4.12. Percentage of area harvested for which: • management guidelines have been completely implemented, and • post-harvest surveys have been conducted to assess the effectiveness of regeneration.	2.1.g. (Sub-national – Australia) Area and percentage of harvested area of native forest effectively regenerated.	4e. Rate of natural regeneration, species composition and survival.

[a]Sources of indicator lists are: ATO (African Timber Organization), 1996; Helsinki Process – Ministry of Agriculture and Forestry, 1993; ITTO (International Tropical Timber Organization), 1998; Montreal Process – Commonwealth of Australia, 1998; and Tarapoto Proposal – Commonwealth of Australia, 1996.

data are collected in the field and recorded spatially (DNRE, 1997a). Data collected as part of the Victorian SFRI project include crown form, cover and width; overstorey height and species composition; conventional stand and tree data; tree profiles, including internal and external stem defect; tree age and growth; tree hollows; as well as fallen (downed) timber and basic understorey species information. The SFRI data are analysed, modelled with other site variables, such as elevation and latitude, and represented in digital format at the regional/FMU level. Remote sensing is used to benchmark the SFRI data against contemporary harvesting and wildfire disturbances.

The SFRI area and volume estimates for forest stand classes, and yield curves for major forest species groups, plus the silvicultural regime appropriate to that forest, are input into the department's forest modelling system – the Integrated Forest Planning System (IFPS). IFPS describes and analyses forest resources and values, including sawlog, residual log, water, wildlife conservation and recreation (DNRE, 1997b). It is Victoria's primary tool for forecasting sustainable yields. It can combine spatial information with textual and modelled information, and can optimize wood flow whilst ensuring a non-declining yield. Attributes such as water yields and constraints on harvesting can be attached to each analysis area, and IFPS used to analyse impacts of alternative forest management strategies, including the effects of more intensive practices such as thinning (Fig. 10.1). IFPS is not used to forecast the sustainable yield of non-wood products such as wild flowers, berries and mushrooms, and game. The sustainable yield forecasts are modelled information at regional level, so comparison with the department's wood and non-wood forest product records (which are stored in temporal databases cross-referenced to FMUs and maps) can be problematic.

The Victorian Code of Forest Practices for Timber Production (DNRE, 1996) specifies that regeneration surveys are to be conducted after timber harvesting or reforestation to determine if those areas are adequately stocked. At the sub-national (State) level, survey procedures are standardized for various silvicultural systems. Data are collected at the FMU level and used to evaluate stocking, competition problems and composition of the new forest stand (DNRE, 1997c). Data are currently stored in temporal format, cross-referenced to maps. In the future, use of SFRI datasets will enable annual area statement updates to be in spatial format, benchmarked against remotely sensed satellite imagery. This will enhance the ability to compare and reconcile area successfully regenerated to area harvested, and area of productive forest.

2 Common indicators in the context of natural forest inventory and forecasting

This section considers three common indicators at levels ranging from the global context to the FMU.

Fig. 10.1. A 30-year-old regrowth forest of *Eucalyptus sieberi* that regenerated naturally following prior harvesting and slash burning in East Gippsland, Victoria. The stand has been commercially thinned with the objective of increasing the growth rate of future sawlog trees. Measures of productive capacity, wood quality, and socio-economic values are relevant to forests managed in this way.

Principle 8 of the Rio Declaration on Environment and Development is 'to achieve sustainable development and a higher quality of life for all people, States should reduce and eliminate unsustainable patterns of production and consumption and promote appropriate demographic policies' (Turner and Pribble, 1996). The focus of this Principle is, as was the Rio Earth Summit, clearly global, yet adoption and implementation of Rio outcomes will enhance the on-ground practice of forestry. However, 'conceptualization of sustainable forest management has out-paced the development of specific on-the-ground practices that will achieve sustainability, and there are many knowledge gaps to be filled' (United Nations . . ., 1998).

Enhancement of our basic knowledge of the science related to indicators of SFM requires government, industry and community commitment, and the development of quantitative or qualitative performance measures and cost-efficient monitoring systems.

2.1 Inventory of natural forests – area of forest available for timber production

Forest resource inventories help decision makers determine if the wood or non-wood resource has potential for development by revealing its apparent abundance, distribution, habitat and condition (Lund, 1998a). Monitoring is the process of observing changes in a resource base over time to determine the ability to meet future demands, and to assess progress of management activities against plans (Lund, 1998b). A forest inventory that uses permanent sample plots can provide the base data for monitoring. Inventory and monitoring of natural forests are generally more complex, expensive, and time consuming than for plantation forests. Natural forests are more variable and generally are more difficult to access.

Inventory methods for collecting wood and non-wood forest resource data can be direct field observations, indirect remote sensing, or a combination. Table 10.2 describes some of the most common approaches to forest inventory (Correll *et al.*, 1997).

Inventories demand considerable resources (labour, technology, energy, transport, etc.) and can be costly (Paivinen and Solberg, 1996). They need to be carefully planned, and where possible linked with other inventories. A challenge for all inventory methods is anticipating and designing for all potential resources and products.

The essential elements of *area of forest available for timber production* as an indicator are:

- reliable, reproducible area statements are needed over pre-determined time intervals;
- measurement to be possible, and meaningful, at levels ranging from the FMU to the national level;

Table 10.2. Some direct and indirect inventory methods.

Direct methods	Indirect methods
Dimensional plots (circular, rectangular, etc.)	Fixed-point/ground-based photography
	Aerial photography and videography
Point sampling (horizontal and vertical)	Satellite imagery
Transect/traverse sampling	Radio telemetry, radar/sonar and other remote sensing systems

- an agreed common approach to describing forests across sub-national/ continental boundaries; and
- standardized area statement reconciliation protocols to account for natural and other losses (including timber harvesting).

This indicator is currently measured using both direct and indirect methods. If measured using remote-sensing techniques and/or recorded spatially, its value to monitor changes in productive capacity is enhanced considerably. It is relevant at the FMU, regional, sub-national and national levels.

Remote sensing is an ideal medium for inventorying and monitoring the extent of natural forests, as well as for many other attributes related to vegetative cover (e.g. biodiversity, watershed protection, soil stabilization and carbon sequestration). The success of remote sensing depends on the type, resolution and scale of the imagery being used. This technology is changing and rapidly improving in sophistication and resolution. It now offers a fairly reliable and practicable method of monitoring changes in forest area. Ferguson (1997) considers that at a national level, it is reasonable to measure the areal extent of forests using this technology every 5 years.

Field inventory has also changed as a result of the availability of new technology (such as global positioning systems and laser-based measuring equipment). Field sampling can now be conducted more efficiently and reliably. Inventory cycles of a minimum of 5 and a maximum of 20 years would seem appropriate and practicable (Ferguson, 1997). Inventories are costly, however, and recently there has been a trend toward multipurpose resource inventories (MRIs). MRIs are data collection efforts designed to meet all or part of the information requirements for two or more resources, goods, products, services (e.g. timber production and watershed protection) and/or sectors (e.g. agriculture and forestry; Lund, 1998b). MRIs can be more cost efficient than single-purpose inventories because data collection efforts can be combined and resources pooled. The Victorian SFRI case study is an example of an MRI designed to collect key non-wood forest data (e.g. on tree hollows for biodiversity modelling) at the same time as timber inventory data.

The use of MRIs is expected to increase as various elements of the UN Conference on Environment and Development (UNCED) and regional processes are implemented. In a worldwide survey of current MRIs, Lund (1998c) found that most:

- were at the national level, with particular focus on environmental and economic needs;
- used remote sensing, with airborne remote sensing (e.g. aerial photography and videography) being the most common;
- used a systematic sample design, with some form of stratification; and
- employed a circular plot configuration (including variable radius or Bitterlich plot design). Many of these were nested to gather a variety of vegetation and other information.

2.2 Inventory of natural forests – extent to which harvested areas are regenerated

Inherent in the concept of ecological sustainability is the notion that the natural mosaic of understorey and overstorey species will be maintained in a regenerating forest. It is generally accepted that, in the absence of human interference, natural disturbances will not adversely affect the natural patterns, processes and productivity of natural forests.

In Australia in the 1960s, Florence (1996) reports there were fears that clear-felling and regeneration by slash burning would adversely affect the ecological stability of forests. These effects included altered natural community patterns, depletion of nutrients, changes to forest composition and productivity, increased potential for endemic disease, and loss of forest wildlife. For these and other reasons, indicators of forest sustainability have always considered changes to forest composition and productivity resulting from timber harvesting.

For forest managers, the main reasons for conducting regeneration surveys include:

- demonstrating that the forest is being renewed;
- obtaining basic information for predicting future forest growth and yield;
- identification of inadequately regenerated areas requiring remedial treatment; and
- evaluation of the regeneration technique used (DNRE, 1997c).

Regeneration success can be measured by the density, distribution, species composition and early growth (quantity and quality) of tree species. Genetic diversity is often not considered. Monitoring regeneration success varies widely in sophistication and intensity between States, forest types and silvicultural systems.

The essential elements of the indicator, *the extent to which harvested areas are regenerated*, are:

- robust yet uniform guidelines for describing 'satisfactory regeneration/stocking';
- consideration of the impact that seasonal conditions can have on regeneration success; and
- development of processes to account for the re-establishment of areas where regeneration has initially failed or is considered inadequate.

This indicator is best implemented at the FMU level, but is equally relevant at the regional, sub-national and national levels. It is ideally suited to being recorded spatially. Once in that format it can be readily compared to, and reconciled against, the 'area of forest available for timber production' indicator.

In the traditional rotation-based management systems, forest development follows a series of well-defined cycles, generally resulting in an even-aged, multi-species forest. The cycle begins with the establishment of a young forest and ends with the harvest of the mature trees. Productivity is measured in terms of the mean annual increment, and sustained yield control is based on the model of the normal forest. Table 10.3 compares some regeneration survey techniques applied to these forests.

A different approach is required for stands of natural regeneration, because it usually occurs in patches of varying height and density. New survey methods have been developed to measure the distribution of species, heights and densities in these uneven-aged multi-species forests. The method developed by Staupendahl *et al.* (1997) uses circular plots of 1.78 m radius and a simple measuring device for rapid height measurement and counting of saplings (Fig. 10.2). As with other regeneration survey methods, the plots containing no regeneration provide as much information about forest structure and species composition as do those plots with data.

Füldner (1995) developed the Structural Group of 4 (SG4) systematic sampling method for estimating tree size data, and spatial distribution of species mingling and size differentiation, in uneven-aged multi-species forests. In addition, unbiased estimates of basal area can be obtained by doing an angle count at each sampling point (Pommerening and Schmidt, 1998).

Table 10.3. Some common regeneration survey methods.

Ground-based surveys	Remote sensing methods
Fixed area plots or quadrats Variable-size plots or distance methods	1 : 500 (large-scale) photography Airborne videography Combination of imagery (e.g. multi-spectral electro-optical imaging scanner – MEIS)

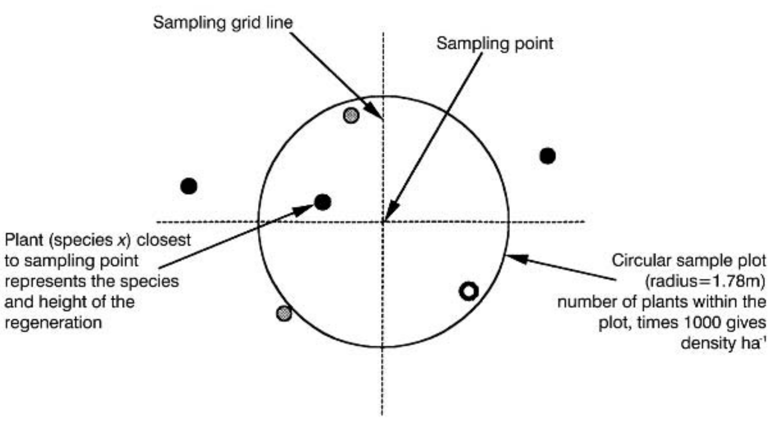

Fig. 10.2. A 10 m^2 sample point for assessing height and density class and species in natural regeneration. In this example species *x* has 3000 plants ha^{-1}.

Attempts to describe the spatial structure of multi-aged, multi-species forest were made by Gadow (1996) and Vanclay (1998). They consider the use of these spatial variables to be an essential precondition for sustainable management of complex forest structures. Structural variables with interpretable reference values also contain useful information about the state of these forests.

2.3 Forecasting in natural forests – level of harvesting for wood and non-wood products compared to the sustainable level

Models are one of the key decision-making tools of forest managers. They are 'abstract representations of the real world that are useful for purposes of thinking, forecasting and decision making' (Buongiorno and Gilless, 1987). Forest management involves consideration of many interrelated variables, and changes to one may profoundly influence the others. The tension and interplay between forest production and conservation is an example. Consequently there are many approaches and factors to be considered in developing and using forest modelling, planning and decision-making tools. Paivinen (1996) compared 12 contemporary forest models and concluded 'a model can only solve those problems in which both data and the data relationships are known'. He considered that wood products and the responses to various management regimes are generally known, and that measuring data and modelling of non-wood products such as forest recreation, berry or mushroom production is problematic. He shares the Barber and Rodman (1990) view that a model is 'the beginning of the process that involves forest specialists, management and the public in the development of plans for the forests. The model is more appropriately used to prevent wrong decisions than for making *right* decisions'.

Forest management planning models need to be capable of handling both temporal and spatial dimensions. In particular the ability to visualize and assess each forecast solution in terms of its spatial context is desirable from both a forest manager's and the public's perspective. Similarly, a critical element in the modelling phase is reconciling the forecast outputs as the scale of planning changes. In Victoria, land units for analysis (with a minimum size between 1 and 5 ha) are developed in ARC/INFO and maintained throughout the modelling cycle. These land units are the model's *common thread*, and ensure the spatial sensitivity, interactivity and accountability of the model as the planning scale changes. These land units, with the remotely sensed data, digital terrain modelling and temporal information, become the modeller's virtual planning space (Lau *et al.*, 1998).

The end-product of forest modelling is generally a sustainable yield forecast determined for a specified set of forest management rules. This forecast can be incorporated into legislation or forest management plans, and used in determining the allowable harvest from a forest. The most complex models have been developed for timber harvesting, generally to manage the balance between timber production, conservation and water. In Australia, simple models have been developed for managing the sustainable supply of non-wood resources such as wild flowers, other flora and wildlife.

The following conditions are required to permit comparison of the *level of harvesting of wood and non-wood products with the sustainable level* indicator:

- products and their units of measurement need to be standardized, or reconcilable, over time;
- the time period for recording needs to be standardized, and record keeping accurate; and
- the process for determining sustainable harvest levels needs to be consistent, or reconcilable, over long periods of time.

This indicator involves the reconciliation of historic, recorded data and long-term, modelled data. As such it is a difficult indicator, but nevertheless fundamental to any expression of sustainability of productive capacity. It is generally not an indicator that can be applied at the FMU level, but is best suited to the regional, sub-national or national level.

The time period is one of the most critical elements in recording, monitoring and modelling the productive capacity of forests. Over time product definitions change, recording and monitoring systems improve, and the science and base datasets for modelling become more sophisticated. Traditionally, sustainable yield forecasts have been calculated over long planning horizons (often in excess of 150 years). However, it may be more sensible to reduce the uncertainty attached to far-distant planning and consider shorter periods within which the scope of possible changes can be foreseen. Ferguson (1997) suggests a planning horizon of 50 years, with discrete planning periods of 5 or 10 years duration.

Data paucity is a limiting factor in modelling forests. In particular the relationships between wood and non-wood goods and services is not well described, and other processes and sets of guiding principles are often used to supplement the analytical planning process (Ferguson, 1997). The advent of geographic information systems and sophisticated spatial analytical tools has enhanced forest modelling. The challenge now is to use these systems to provide transparency to the forest management planning processes.

3 Recommendations for further research and development

Natural forests are more complex, variable and difficult to access than plantations; similarly their management is more complex.

A considerable knowledge base already exists for most of the world's natural forest types. Much of this knowledge relates to traditional wood values and within-forest management, with significantly less known about the non-wood values and the relationships between forests and the adjoining lands. However, for the three common indicators of productive capacity, there are a number of enhancements that can be made using this knowledge base as a platform.

The *area of forest available for timber production*, particularly if related to species, is the basic FMU. It is commonly represented as a map, sometimes, but not always generated from digital data. The essential elements of this indicator are use of a common definition of forest, and consistency of measurement. Further research and development is needed into:

- remotely sensed signatures (particularly reflectance characteristics) of complex forests (such as multi-aged, multi-species forests). Radar and laser altimetry methods are probably the most deserving of further research;
- remote sensing techniques which enable the combination of different technologies, thereby bringing the respective strengths of each technique to the combined product. The combination considered to have the most potential is digital (radar) data with optical (video) imagery; and
- enhancements to measuring equipment (particularly the use of laser technology) and position-determining devices/systems to enable more accurate forest attribute measurement from known ground truthing sites.

Advances in forest area mapping will be coupled with similar research and development needs for other indicators, in particular the use of remote sensing for biodiversity mapping of forest type, age class and fragmentation.

Remote sensing will also be a powerful technique applied to the common indicator, the *extent to which harvested areas are regenerated*.

Harvested, and subsequently regenerated, areas are generally represented in map form at the FMU level. Hence the techniques applicable to forest area can be applied to determine the area harvested (where the harvesting system is more intense than single tree or small group selection), and possibly the area regenerated. The application of remote sensing techniques will be limited by the level of resolution (pixel compared to forest gap size) and the reflectance characteristics of young forest species, particularly the commercial species.

Implicit in describing the area regenerated is the ability to define 'satisfactorily regenerated/stocked' forest. Ground survey is the main technique applied to seedling-sized regeneration. Research is needed into both these elements of this common indicator.

In particular, research and development is needed into:

- remotely sensed signatures for harvested and regenerated forest areas, and remote sensing techniques combining different technologies (for details, refer to our earlier recommendation on 'area of forest' indicator). Radar, laser altimetry and videography techniques are considered to have the most potential;

- efficient, rapid ground survey techniques and sampling designs for seedling-sized regeneration. Innovations akin to the techniques described in the chapter for multi-species; even-aged, uneven-aged and multi-aged forests are required; and
- forest stand dynamics, and the development of indices of variation and diversity, contagion, species mingling and size differentiation for a range of forests. As a result the level of satisfactory regeneration or stocking can be better defined and understood. Sufficient research may have already been completed in a number of forest types, so only the development of standard definitions at the regional or FMU level may be required.

The common indicator, *level of harvesting for wood and non-wood products compared to the sustainable level*, contains elements of record keeping and forest modelling. Wood and non-wood harvesting records are usually registered at the FMU level. Forest modelling is generally conducted at the strategic regional or sub-regional levels. Reconciliation of these two elements is critical to this indicator.

The use of spatial data in forest modelling has enabled the application of very site-specific rules, and tailoring of forest management regimes to particular forests and sets of products. Linking forest modelling and analysis techniques to remotely sensed data layers has been an important breakthrough for forest monitoring and management.

Considerable knowledge already exists about the relationship between wood products and various management regimes, but research and development is needed into:

- the relationship between wood and non-wood values. Strong links exist between this requirement and the research and development needs for the biodiversity (flora and fauna habitat management) and the water indicators. There will be many opportunities for collaborative research;
- modelling techniques that enable the scale of planning to change without compromising the resolution or accuracy of the forecast estimates. This technique will be required to demonstrate that sustainable yield forecasts at the strategic (regional) level adequately address planning constraints at the FMU level (for example, low-level timber harvesting within defined water supply catchments);
- develop modelling techniques that express both a long-term management context and medium-term flexibility. Forest management is by necessity long term, but wood and non-wood product requirements, and forest policy objectives, can change over shorter time periods. The forest modelling capability needs to be developed so that one longer-term model (expressing long-term management objectives) can run parallel with a series of short-term models (expressing short-term changes to product requirements); and

- develop processes within the indicators of sustainability to enable information from models (developed from remote, direct or indirect sources) to be merged with information from representative point samples (e.g. coupe harvest information).

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Indicators for Sustained Productive Capacity of New Zealand and Australian Plantation Forests

11

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Australia and New Zealand are committed to the concept of sustainable forest management. Extensive portions of native forest in each country are managed primarily for conservation purposes. Substantial investment has been made in both countries to establish planted production forests, which together comprise 54% of the global *Pinus radiata* (radiata pine) estate. Financial criteria will determine the extent of planted production forest in the future; but environmental legislation has been enacted to sustain the productive capacity of existing ecosystems. Indicators of forest ecosystem productive capacity should include assessments of trends in growth and yield over time, and be linked to changes in site quality. Detecting change in forest productive capacity due to changes in site quality between rotations remains difficult due to confounding effects of silviculture, genetic stocks, disease and insects, and climate variability and change. Additional site-specific technical information and predictive models are required to develop indicators of sustainable forest management practices for environmental monitoring in plantations at the management unit level. Cost-effective monitoring will involve application of indicators at varying spatial and temporal scales.

1 Introduction

Australia and New Zealand participated in the 'Montreal Process' with ten other countries to develop a common, international framework for describing and assessing progress towards sustainable forest management at the national level. In 1995, the Montreal Process resulted in the signing of the 'Santiago Declaration' which endorsed a comprehensive set of criteria and indicators (C&I) for forest conservation and sustainable management for use by national governments (Anon., 1995). In the past 5 years, the Montreal Process Technical Advisory Committee has continued work on definitions of terms, rationale for C&I, considered methodology for measuring indicators, and evaluated means for aggregating regionally collected data at the national level.

The seven criteria and indicators endorsed by the Montreal Process countries are not substantially different from frameworks for defining the economic, environmental and social goals or values important to society proposed by other international agreements, such as the Helsinki Process (Anon., 1994). The forest products and pulp and paper industries of many countries have also developed versions of sustainable forestry initiatives, prompted in part by the United Nations Conference on Environment and Development (UNCED) in 1992 (UNCED, 1992; Brand, 1997). For example, the American Forest and Paper Association developed Sustainable Forestry Principles and Implementation Guidelines (AF&PA, 1995), to which members must adhere to retain membership (Lucier and Shepard, 1997). In general, six criteria have been common to all international agreements, namely biodiversity; forest health; forest productivity; soil and water conservation; contribution to global carbon cycles; and socio-economic values. The Montreal Process has extended beyond this core with a seventh criterion for a legal, institutional and economic framework for forest conservation and management. In conclusion, there is general international agreement about the definition of sustainable forest management (SFM).

The challenge facing the international forestry sector is to develop the scientific basis and technical knowledge required for applying C&I of sustainable forestry to adaptive management systems at the management unit level. In practical terms, foresters need tools, guidelines or management systems to enable them to turn high-level, subjective sustainability goals into outcomes that can be measured quantitatively.

The objectives of this chapter are to review the state of knowledge related to Montreal Process Criterion 2, 'Maintenance of productive capacity of forest ecosystems', with specific focus on applicability to planted production forests; to evaluate the scientific basis for detecting changes in productive capacity at the forest management unit (FMU) level; to discuss linkages between Criteria 2 and 4, 'Conservation and maintenance of soil and water resources'; to evaluate the applicability of current growth models to describe the functional relationship between forest productivity and soil-site quality; and to make recommendations for research required to develop scientifically based indicators for adaptive forest management applied on a site-specific basis.

2 C&I of SFM

The criterion 'maintenance of productive capacity of forest ecosystems' is essential because society depends on forests for a variety of goods and services. The indicators of forest productive capacity at the national level include measures of the area of forest land, including naturally regenerated and planted native and exotic forests; an assessment of growing stock and growth rate; and an assessment of the sustainability of annual removals of forest products. This criterion is potentially a measure of the resilience of national forests to disturbance and stress, and the productive capacity of forests is closely related to conservation of soil and water values. An assessment of forest growth rates may be required to determine changes in forest productive capacity. However, analysis of trends in forest growth and yield over time must be able to distinguish between changes in forest area and changes due to soil quality, silviculture and genetic stock.

Australia and New Zealand have made substantial investments in plantation establishment as well as conservation of native forests for non-timber values. Both countries have placed great emphasis on developing means of quantifying SFM at national and management unit levels.

2.1 Australian forest estate

Australia is a net importer of forest products both in volume and value. In 1997/98 the annual trade deficit in this sector was Aus$1.5 billion. About 20% of Australia's land area, or 156 million ha, is occupied by native forest, which is defined as plant communities dominated by trees with an expected stand height exceeding 2 m and potential crown cover of 20% (McLennan, 2000). Only 3% is closed forest with more than 80% crown cover, while open forest (50–80% crown cover) dominated by eucalypts occupies 25% of the forest estate. About 72% of native forest is publicly owned, with about 13 million ha being managed for multiple use including wood production. Significant areas within multiple use forest are managed primarily for conservation, and this area is increasing during the implementation of arrangements known as Regional Forest Agreements. There is a trend toward 'zoning' of the wood production forest into areas to be conserved, areas to be lightly logged where conservation is given priority, a general management zone with a balance between conservation and production, and areas more intensively managed for wood production.

Fig. 11.1. Even on productive farmland, tree planting can be a valuable way of diversifying and stabilizing income, as well as creating valuable habitat. Growing of high value hardwoods for sawn timber is expanding. Plantation development is also being stimulated by the need to rehabilitate degraded land, and by proposed trading of carbon credits.

In 1998 there were about 1.2 million ha of plantations in Australia, of which about 55% were radiata pine (*Pinus radiata* D. Don), 21% other softwoods and 24% eucalypts (McLennan, 2000). Eucalypt plantations, mostly for pulpwood, are increasing. During 1997/98, 53% of total roundwood removals came from plantation forests. There is a vision shared by industry and State and national governments to triple the size of the plantation estate by the year 2020 (Fig. 11.1). Most of the expansion will be on cleared farmland.

In 1988 the Australian Government initiated a National Forest Inventory (NFI) which has issued a report including a description of the forest resource, its use and management with an examination of the social forces framing public opinion (NFI, 1998). A National Plantations Inventory has also been initiated to provide a comprehensive information base to enable forecasts of regional and national wood flows from the plantation resource (NFI, 1997). The Australian Bureau of Agricultural and Resource Economics (ABARE) produces a quarterly review, *Australian Forest Products Statistics*, which summarizes production and consumption, imports and exports of sawn wood, processed wood products and pulp and paper products; and changes in the resource base by ownership, forest type and plantation species. Regular collection of statistics such as these are vital as indicators for monitoring productive capacity of wood production forests at regional and national levels. However, much less is known of productive capacity of forests used predominantly for grazing or other non-forestry pursuits.

2.2 New Zealand forest estate

The total land area of New Zealand is 27.0 million ha. Native forests historically covered 80% of the country; and only 15% of rich, lowland forest types remain. Currently 24% (6.4 million ha) of the land area carries natural, indigenous forest (based on the National Forest Survey Data Base, conducted in 1945–1953 and updated in 1974). The majority of native forests are protected for conservation purposes under a combination of legislation and voluntary measures, including The Conservation Act 1987, The Forests Amendment Act 1993 and The New Zealand Forest Accord 1991. As a result, only 2% (130,000 ha) of native forests are available for timber production, and these must be managed sustainably according to an established code of practice (Ministry of Forestry, MoF, 1993). Fifty-one per cent of the land area (13.8 million ha) is pasture and arable land; 19% (5.1 million ha) is other non-forested land; and 6% (1.6 million ha) is planted ‘production’ forest (New Zealand Forest Owners Association, NZFOA, 1997; Ministry of Agriculture and Forestry, MAF, 1998).

Planted production forests are composed of 91% radiata pine, 5% Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), 2% other exotic softwoods, and 3% all exotic hardwoods (including eucalypts) (MAF, 1998). The area of land planted with exotic species for conservation and protection (e.g. stream bank stabilization and erosion control) purposes has not been quantified with any precision. Planted production forests supplied 98.7% of the total roundwood removals from New Zealand forests in 1994 (MoF, 1994). Production forest exports were projected to account for 12% of total New Zealand exports in 1997 (NZFOA, 1997). The total area planted in 1996 was 111,800 ha, of which 83,600 ha was new planting, and 28,200 ha was restocking (MAF, 1998). Thirty-six per cent of new planting occurred on improved pasture, 50% on unimproved pasture, and 14% on land previously occupied by scrub land cover. About 67% of the radiata pine estate is intensively tended (pruned to a height of at least 4 m) and managed primarily for solid wood products. In the year ended 31 March 1997, the area-weighted average clear-fell age of radiata pine was 27.8 years (MAF, 1998).

New Zealand maintains a National Exotic Forest Description (NEFD), which is an inventory of the planted production forest resource. Regular NEFD reports have been produced since 1983, and include the basic NEFD (MAF, 1998) as well as NEFD yield tables (MoF, 1996). The NEFD report summarizes the area of planted forest by age, species group and general tending regime for each territorial local area authority. As such, the NEFD is well suited to quantify the spatial extent and changes in growing stock associated with indicators of the criterion for 'maintenance of productive capacity of forest ecosystems'.

2.3 Plantation perspective on SFM

New Zealand climate and soils are favourable for achieving relatively fast growth for radiata pine and several other exotic plantation species. For example, on good sites, radiata pine mean annual increment (MAI) can average 30 m^3 ha^{-1} and be as high as 50 m^3 ha^{-1} (Maclaren, 1993). The Stand Growth module of STANDPAK (Whiteside, 1990) predicts MAI at typical harvest age of about 28 years from 22 to 29 m^3 ha^{-1} with 300 stems ha^{-1} final crop stocking across the seven major forest regions of New Zealand (Fig. 11.2).

Fast growth of exotic plantation species allows New Zealand to achieve essentially all of its forest products export earnings from 6% of its land base and, in a sense, reduces economic pressure to harvest native forests. The relatively large conservation estate enables satisfaction of some of the environmental criteria of SFM at the national level (e.g. biodiversity) without requiring all criteria to be satisfied in every management unit. As a result, the plantation estate can be primarily managed on a commercial basis with profit as a prime objective. Similar arguments can be applied to Australia where expansion of the plantation estate will reduce the need to harvest native forests.

As discussed by Bigsby (1995) and Richardson *et al.* (1999), the sustainability of the extent of plantation forests will be primarily determined by

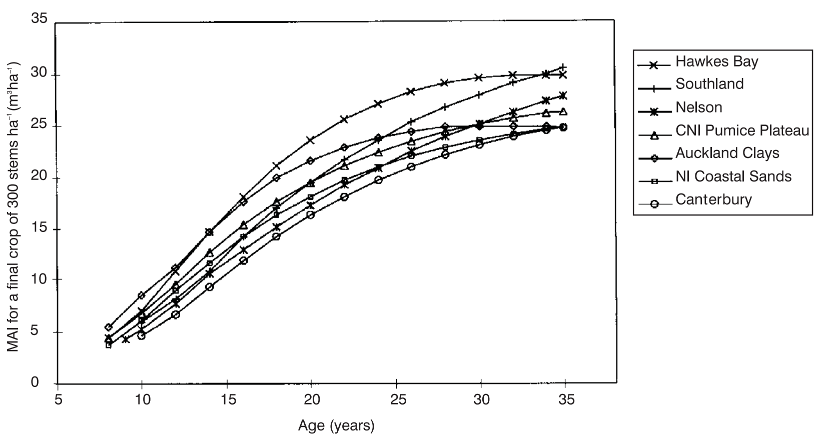

Fig. 11.2. Comparison of regional forest productivity (MAI) for radiata pine in New Zealand, as predicted by seven regional models in STANDPAK (West *et al.*, 1982), assuming genetic stock with GF factor 16, stand pruned to 6.4 m, thinned to 300 stems ha^{-1} final crop stocking at about age 7 years. Regional functions were used for height, basal area growth and volume.

financial criteria. The current rate of land-use conversion by new forest plantings on pasture and scrub lands, 83,600 ha in 1996, is taking place because forestry is more profitable than other alternatives. Bigsby (1995) points out there is no reason why the New Zealand public would be more concerned about reductions in the plantation estate than other shifts in commercial land use, such as pastoral agriculture or horticulture, if such changes reflect financial conditions.

However, New Zealanders are concerned about sustained biological capacity of ecosystems, and legislation embodied in The Resource Management Act 1991 requires that all land-use practices sustain the productive capacity of the soil. This principle has been extended to the plantation estate in the *Principles for the Sustainable Management of New Zealand's Commercial Plantation Forest*, which have been endorsed by major stakeholders in the forestry sector. These principles for sustainable land management strengthen the importance of linkages between criteria for maintaining productive capacity and conservation of soil quality. In Australia, the Commonwealth and all State and Territory Governments have signed a National Forest Policy Statement which provides a policy framework for future management of Australia's public and private forests and outlines a vision for their ecologically sustainable management (Commonwealth of Australia, 1992). This policy is being implemented by negotiating Regional Forest Agreements for the long-term management and use of forests in particular regions.

In the plantation context, lands may be managed intensively for wood production with substantial silvicultural inputs (e.g. fertilizer, insecticide and site preparation) if economic analysis supports such decisions. In plantations, industry would analyse trends of various indicators in a different context than for extensively managed forests. For example, extensively managed natural forests with low levels of inputs may require harvest utilization standards designed to achieve balanced site nutrient budgets over the long term, taking into account factors such as nitrogen fixation and atmospheric and weathering inputs. In planted production forests, harvest utilization recommendations for intensively managed forests could permit high nutrient removals assuming replacement by fertilizer or other sources (e.g. legumes). Decisions about the degree of organic matter and nutrient removal, soil compaction, and other measures of soil quality would be based on economic analysis of the relative costs of modifying harvesting equipment and delimbing operations, replacement of organic residues, fertilizer and the degree of growth loss. Such cost-benefit analysis would determine whether the cost of retaining nutrients and organic matter during harvest was greater than the cost of ameliorating soil fertility and correcting growth loss with fertilizer or other soil amendments.

In Australia, consideration of factors such as these have led to alternative strategies for the management of plantations on deep sands. These soils are characterized by low levels of nutrients and organic matter, which is a prime determinant of chemical and physical properties. Evidence of decline in growth of successive rotations of pine (Keeves, 1966) was subsequently linked to changes in a range of soil properties (e.g. bulk density, nutrient status and moisture availability) between rotations (Squire *et al.*, 1985). In Victoria, management of these soils has placed strong emphasis on limiting or avoiding slash burning to reduce loss of organic matter and nitrogen (Flinn *et al.*, 1975; Squire *et al.*, 1985), while in South Australia the initial emphasis was on more concentrated usage of fertilizers and weed control (Woods, 1980). Utilization and establishment practices aimed at conserving organic matter and site nutrients are now commonplace throughout the country.

2.4 Assessing changes in forest productive capacity

Montreal Process C&I related to 'maintenance of productive capacity of forest ecosystems' should include an assessment of trends in growth and yield over time. For plantations, such assessments must identify if management practices are changing site productive capacity, thus serving as a link between Criteria 2 and 4 'conservation and maintenance of soil resources'. Methodology to determine changes in plantation growth rates from one rotation to the next logically includes elements of growth model comparisons, continuous forest inventory using permanent sample points (PSPs), and experimental research trials.

Use of growth models to determine changes in growth between rotations would involve modelling growth for each rotation mathematically and comparing model parameters statistically for significant differences between rotations. The precision of comparisons would be expected to increase with the degree of repeated sampling at the same location and control over independent variables affecting growth. Assessing changes in site productive capacity from rotation to rotation is not simple given interactions among factors affecting forest productivity, including soil quality, climate variability, silviculture (e.g. stocking, thinning, pruning, weed control, fertilizer), disease and insect attack, site preparation operations and genetic stock (Fig. 11.3). Assessment procedures to identify changes in site productivity due to changes in soil quality must first be able to 'control' or explain variation in tree growth due to non-soil related factors such as silviculture and genotype.

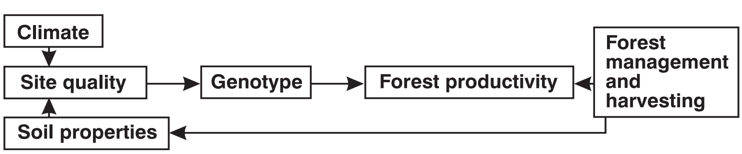

Fig. 11.3. Conceptual diagram of the relationship among the factors affecting forest productivity (after Dyck and Bow, 1992).

Burger (1994) discussed a concept developed by Switzer (1978) using hypothetical logistic growth curves of biomass accumulation over time to describe how species biological potential, environmental stress and site carrying capacity affect the rate and maximum amount of biomass accumulation. Silvicultural factors and genetic improvement are designed to increase both biomass accumulation rate (curve slope) and maximum production (curve asymptote). Conversely, site degradation (e.g. soil compaction, reduced nutrient availability), weeds and increases in climatic stress decrease production rate and site productive potential. This conceptual approach to growth curve analysis illustrates the need for long-term data to distinguish accurately among changes in growth rate and carrying capacity.

Growth models, commonly used by forest industry for radiata pine management in New Zealand, are predominantly based on state-space models first developed by Garcia (1984, 1994) which comprise a set of stochastic differential equations. These models predict the rate of change of stand parameters of top height, standing basal area and stems per hectare over time on the assumption that future development is determined by the 'state' values of the stand at some point in time (Goulding, 1995). Thus, state-space models predict that all stands with identical state values will develop identically over some period of time. Most Australian States, Territories and major forestry companies have a variety of models for forecasting the growth of their plantations.

In New Zealand, regional differences in growth and yield are accounted for by seven models for radiata pine. These include models for Auckland Clays, North Island Coastal Sands, Central North Island Pumice Plateau, Hawkes Bay, Nelson, Canterbury and Southland which are mostly based on several state variables such as age, stocking, basal area, top height and site occupancy. However, several regional models include modifiers for predicting growth for different fertility levels, including modifiers for Auckland Clays based on foliar phosphorus levels, and other models include nitrogen modifiers for North Island Coastal Sands, Golden Downs Forest, and the Pumice Plateau (Goulding, 1995). The effects of early silviculture, particularly the effects of very heavy pruning and thinning, are predicted by the model EARLY (West *et al.*, 1982). The EARLY model allows users to set basal area increment to suit local site conditions and hence is used on a national basis.

Mean annual increment predicted for stands from 8 to 35 years age for seven New Zealand regions indicates substantial differences in growth curve shape and MAI at average harvest age of 28 years (Fig. 11.2). These patterns are presumably related to differences in site, but recent growth modelling studies have not explained how site affects growth. For example, although the seven regions represent distinct differences in soil parent material, climate and physiography, regional differences in tree diameter growth were generally not accounted for by site, climate and nutrition information available for 271 PSPs studied by Gordon and Lawrence (1997). In their study, stand-related independent variables explained 48–76% of the variation in radiata pine diameter increment in stands aged at least 15 years which received no additional thinning and pruning at older ages. Despite substantial differences in soil and climate, only Canterbury (rainfall, nitrogen) and Southland (rainfall) regional models included significant site-related variables.

Recent studies by Woollons *et al.* (1997) in the Nelson region of New Zealand and Snowdon *et al.* (1998, 1999) in the southern tablelands of New South Wales, Australia, indicate that environmental data (e.g. seasonal rainfall and soil information) can increase the precision and accuracy of growth projection models. Similar results have been reported for the Hawkes Bay region of New Zealand (Woollons *et al.*, 1998). These studies support the recommendation that the form of environmental variables (e.g. seasonal versus annual rainfall) used in growth models must be related to tree biology.

The results of 14 studies from Australia, New Zealand and South Africa evaluating soil–site relationships for radiata pine indicated soil–site factors have explained between 33% and 90% of the variation in growth (summarized by Richardson *et al.*, 1999). Highest model precision has generally been from ‘local’ studies or research trials with relatively tight control of independent variables affecting growth. For example, Benson *et al.* (1992) developed a model based on foliar nitrogen and water that explained 90% variation in growth in the Biology of Forest Growth project near Canberra, Australia. Regional studies in areas with large environmental and site gradients generally report strong site–growth relationships. For example, Turvey (1983) developed yield curves plotting total volume over age which showed strong differences in curve shape and maximum MAI for seven soil types in Gippsland, Victoria. Regional studies relating growth to site variables typically have explained about 60% variation in growth (e.g. Hunter and Gibson, 1984), presumably because of failure to measure key ‘driver’ variables, less precision and accuracy in independent variable measurement, and greater variation in growth.

An assessment of existing growth and yield models indicates poor ability to quantify functional relationships between tree growth and environmental variables with the degree of precision required for making site-specific management decisions for species selection, fertilizer application and harvesting equipment based on soil physical and chemical properties. That is, existing models fail to account for the relationship between tree growth and soil quality *after* controlling for the influence of stand ‘state’ variables. It is reasonable to assume this failure is due to lack of control over site-related environmental variables, rather than lack of relationship. Furthermore, lack of understanding of functional relationships between site variables affected by forest management and site productivity limits development of indicators of SFM. Progress towards developing indicators of change in forest productivity that are sensitive to site degradation requires new site-specific research.

Detecting changes in site quality between rotations will be difficult because of differences in silviculture and genotype, and interpretations may be complicated by introduced diseases and insects. One advantage of the state-space approach suggested by Snowdon (1997) to detect productivity changes between rotations is the ability to compare stand development among stands with similar values of stand-related state variables, thus isolating the effects of changes in site quality. Identifying stands with adequate similarity in stand-related state values remains problematic, but is the key to application of this technique for detecting productivity differences through time.

2.5 Application of C&I to adaptive forest management

Research conducted across a range of site fertility in New Zealand is leading to the development of site-specific recommendations for harvest residue management, based on relationships between soil nutrient availability and tree nutrition and growth determined with a high degree of experimental control (e.g. Smith *et al.*, 1997). Similar case study research can lead to ecologically sustainable SFM practices developed for application at the management unit through the conceptual approach illustrated in Fig. 11.4. However, additional site-specific research is needed to develop the technical, cause-and-effect knowledge to underpin codes of forest practice, 'local' management prescriptions, environmental indicators for monitoring the consequences of management, and interpretations for adapting management.

Environmental indicators of SFM should have the following attributes:

- easy to measure; cost-effective; accommodate changing conditions (e.g. time, crop age); scientifically sound and based on functional ecological

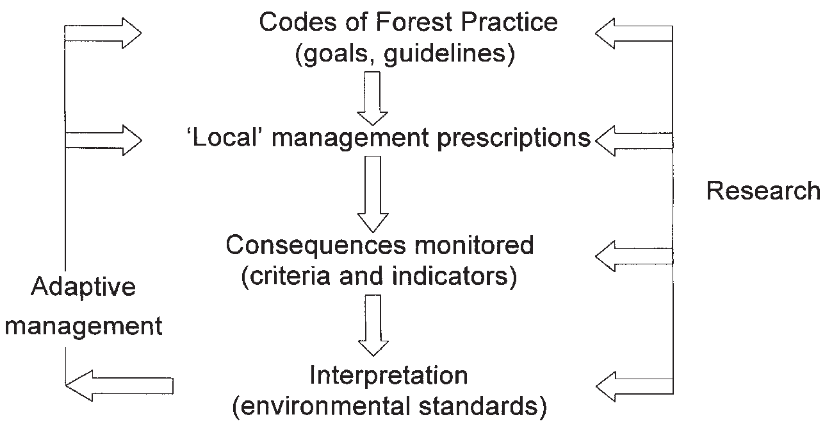

Fig. 11.4. The linkages between research and the adaptive management process in achieving ecologically sustainable forest management (after Smith and Raison, 1998).

relationships; forest ecosystem specific, yet able to be 'scaled-up' (e.g. using spatial statistical techniques and Geographic Information Systems, GIS);

- integrative of ecosystem functional relationships (e.g. many indicators chosen to represent selected, key ecosystem processes; or fewer, key indicators which integrate across the entire ecosystem); and related to management goals or values.

We recommend environmental monitoring be applied at varying spatial and temporal intensities. *Cost-effective* monitoring could be achieved by varying monitoring intensity as follows: all sites could be monitored for operational compliance; a limited number of sites could be monitored to determine effectiveness of Best Management Practices with site-specific indicators; and very few 'benchmark sites' (on representative sites) could be monitored intensively to validate research recommendations and to adapt management practices (Fig. 11.4).

3 Conclusions

Australia and New Zealand have made substantial contributions to developing the concept of SFM. Extensive proportions of native forest in each country are managed primarily for conservation purposes. Substantial investment has been made in both countries to establish planted production forests. Australia currently contains around 19% and New Zealand around 35% of the global radiata pine estate. Financial criteria will continue to determine the extent of planted production forest in the future. In plantations, the concept of SFM practices should be defined in the context of economic profitability of the forestry enterprise compared to alternative land uses. We acknowledge the potential for site degradation to take place in the absence of adequate early-warning indicators of declines in soil quality, and poorly defined relationships between soil quality and stand productivity. Environmental legislation has been enacted in New Zealand to reduce negative environmental effects of land management and sustain the productive capacity of ecosystems. Indicators of forest ecosystem productive capacity should include assessments of trends in growth and yield over time, and be linked to changes in site quality. Detecting change in forest productive capacity due to changes in site quality between rotations remains difficult due to confounding effects of silviculture, genetic stocks, diseases and insects, and climate variability and change. Additional site-specific technical information and predictive models are required to develop indicators of SFM practices for environmental monitoring in plantations at the management unit level. Cost-effective monitoring will involve application of indicators at varying spatial and temporal scales.

Acknowledgements

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12 Indicators to Guide Management for Multiple Forest Use

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Within the lifetime of the latest generation of trees, important changes have occurred in the utilization of many forests, as well as in their physical and chemical environment – especially if they are growing in polluted areas. Current management for forest production and nature conservation often does not take these changes into account. In this chapter, an improved concept for multiple (multifunctional) forest use, with guidelines for its implementation, is presented. A classification scheme for indicators to describe and evaluate changes in ecosystems is outlined, and an extended critical loads concept is suggested.

The concept of multifunctional forest use includes four primary elements of forest functioning: regulation of matter and energy, habitat conservation, production and use of resources, and cultural and social role. It is based on the ecological principles of minimization of waste, symbiosis, diversity, and a high level of elasticity, stability and resilience. It aims to provide a basis for sustainable forest management.

The scheme proposed to classify indicators is built on integrative ecosystem theory, based on processes in forest ecosystems, and considers a hierarchy of different indicators. Simple analytical indicators (measurable, non-evaluating, i.e. providing little basis for assessing the reasons for or importance of change), compound indicators (measurable, non-evaluating), system indicators (cannot be measured directly, non-evaluating) and normative indicators (evaluating) are described.

The extended critical loads concept presented here relates to significant changes in the functioning and use of forests. It consists of critical amounts of material input (loads), critical levels of forest operations,

(eds R.J. Raison, A.G. Brown and D.W. Flinn)

critical states and functions, and critical amounts of loss from terrestrial ecosystems. This concept includes site properties as an important factor affecting ecosystem and landscape responses and considers time as an important variable.

Adherence to the static evaluation of site factors either in forestry or in nature conservation is unlikely to be useful. The concept presented in this chapter provides a framework for the development and evaluation of indicators which reflect the dynamic nature of site factors, and change in the functioning and vitality of forest ecosystems.

1 Introduction

A major function of forests remains the production of wood which, as a renewable raw material, is gaining increasing importance in a world with a rapidly expanding population. Despite this importance, the supply of wood per capita is likely to decrease in future because of the disparity between the demand for wood and its supply, which is affected by changing climatic and non-climatic factors (Dixon *et al.*, 1994; Burschel, 1995; Schulte-Bisping *et al.*, 1999). Forests are an integral part of the landscape and they have a multifunctional role in the environmental, ecological, economic and social well-being of a society.

Forest ecosystems are open systems which exchange matter, energy and genetic information with other systems. Thus they are susceptible to change from external inputs (loads), particularly those associated with forest use and climate. Loads may arise from exchanges of energy and matter, the establishment of non-native species (e.g. Kilian, 1998; Noble *et al.*, 1999) and other forest management practices. Ensuing alterations in the structure and functioning of an ecosystem may be irreversible if the internal capacity of the system to buffer or repair effects induced by the loading is exceeded. It has therefore been suggested that input–output budgets for elements are a useful diagnostic indicator of sustainable forest management (SFM, Ranger and Turpault, 1999).

During the lifetime of the latest generation of trees, important changes have occurred in the utilization of many forests as well as in the physical and chemical forest environment, especially in polluted areas (Sperber, 1995). Consequently forest ecosystems have changed, and any attempt to define site factors as static characteristics (e.g. Pfeil cited in Hasel, 1977) may be futile. Ulrich (1991) concluded that these ecosystem parameters can be rapidly altered by humans, directly or indirectly.

An important task will thus be to develop strategies for managing the utilization of forest resources which accommodate the dynamic nature of site characteristics. Standards with which to evaluate forest utilization (considering the four main functions of forests, discussed below) and consequent changes will be required. These standards must not be limited to production or tree growth, but must cater for other equally important functions of forests. Furthermore, the loads imposed by the utilization of forest resources on neighbouring terrestrial and aquatic systems and groundwater should also be included. Utilization of forest resources should be sustainable and environmentally sound. This will be possible, however, only if forest management is site-compatible. Whether the standards can be used as workable indicators will depend upon the desired goals and economic factors. Nevertheless, an indicator system which goes far beyond existing approaches by recognizing the multifunctional nature of forests will be required.

Objectives of this chapter are: (i) to provide a theoretical background discussion on future multifunctional forest use; (ii) to present a classification of indicators that can be used to describe and evaluate changes in forest ecosystems; and (iii) to outline the indicators needed for a wide-ranging critical loads concept for forest ecosystems. Examples of this concept are provided by indicators to assess changes in soil-based parameters, considering critical levels of nutrients in soils in relation to critical loads on the system, critical levels of forest operations and their effect on soils, critical levels of soil parameters and their functioning, and critical values of nutrient losses relative to requirements in the system and 'off-site' effects.

2 Multifunctional forest use

The concept of multifunctional forest use includes four primary forest functions: regulation of matter and energy, habitat conservation, production and use of resources, and cultural and social roles. In order to achieve sustainability of these functions, forest use should follow a number of ecological principles. Some of these are: (i) minimization of waste; (ii) avoiding any negative impacts of resource utilization on productivity; (iii) maintaining diversity; (iv) achieving a high level of elasticity, enabling an ecosystem to buffer deviations caused by climate, growth or utilization in such a way that the organisms of the system are not endangered; (v) maintaining stability, whereby an ecosystem oscillates around a steady state (Ulrich, 1987); and (vi) maintaining resilience, enabling an ecosystem to return to its original state even if the limits of elasticity are exceeded (Ulrich, 1987).

Multifunctional forest use requires site-compatible, sustainable and environmentally sound management. This is often achieved by zoning the forests into units that have differing management objectives and associated management practices. An example is shown in the colour map on the reverse of the frontispiece. Four attributes of multifunctional forest use are described below (for details refer to Beese, 1996).

2.1 Regulation of matter and energy in forest ecosystems

Multifunctional forest use preserves the internal cycles of an ecosystem and reduces material loads on neighbouring systems. This is achieved by: (i) minimizing all those processes that may disrupt matter and energy transformations; (ii) synchronizing the decomposition of dead biomass with the transformation and uptake of matter (especially nutrients) by living biomass, and the release and accumulation of energy by an ecosystem; and (iii) minimizing irreversible degradation of soil and other environmental resources.

This attribute has similarities to the Montreal Process Criterion 4 (conservation and maintenance of soil and water resources) and Criterion 5 (maintenance of forest contribution to global carbon cycles) (UN, 1995).

2.2 Habitat conservation and functioning

Multifunctional forest use is expected to preserve the elements of biological diversity and associated biological activity which will lead to increased elasticity and resilience of a forest ecosystem. This is attained by: (i) maintaining or increasing floral and faunal diversity of forests in temporal and spatial scales; (ii) establishing mixed-aged and mixed-species forests (if appropriate); (iii) protecting or improving soil structure, chemical status and biological functioning, especially in polluted areas; (iv) establishing conservation areas with minimal resource exploitation activities; and (v) minimizing inputs of toxic substances to forest ecosystems.

These aspects of habitat conservation are emphasized in the Montreal Process Criterion 1 referring to the importance of conservation of biological diversity, and Criterion 3 relating to the maintenance of forest ecosystem health and vitality.

2.3 Production and use of resources to meet economic and ecological needs

Multifunctional forest use will improve efficiency in the use of resources needed for production by: (i) reducing losses of matter and energy by better sharing resources between various components of an ecosystem (e.g. recycling of ash; Kahl *et al.*, 1996); (ii) balancing element losses or correcting nutrient deficiencies (e.g. use of fertilizers to offset exported nutrients; Mackensen and Fölster, 2000); (iii) reactivating self-regulating soil and plant processes (e.g. reintroducing earthworms in amended acid soils; Geissen *et al.*, 1997); (iv) optimizing forest and soil management practices (e.g. minimizing disturbance during harvest); and (v) prudent management of plant, animal and environmental resources (e.g. leaving the slash on the site after harvest; Flinn *et al*., 1979; Takahashi, 1995).

The maintenance of productive capacity of forest ecosystems is an important criterion of the Montreal Process, and indeed of other international processes describing criteria and indicators (C&I) of SFM.

2.4 Cultural and social role of forest ecosystems

Multifunctional forest use may stabilize rural societies by providing social and cultural benefits to the whole population. This can be achieved by: (i) optimizing the use of renewable raw materials produced in the forests (wood and other forest products); (ii) preserving jobs and income of populations depending upon forests for their livelihood; (iii) maintaining man-made rural landscapes; (iv) increasing the social role of forests in the daily life of the people; and (v) conserving cultural heritage sites found in forests.

The Montreal Process Criterion 6 emphasizes the significance of the maintenance and enhancement of long-term multiple socio-economic benefits to meet the needs of societies.

3 Indicators and their classification

The concept of multifunctional forest use presented above is complex, and requires extensive and diverse information and quantitative data to provide the basis for management decisions and action. The complexity and interactions between the various criteria involved potentially require many indicators to ascertain and evaluate the existing status of ecosystems, and their likely future development. The 'right' indicators (or surrogates of indicators) are therefore required so that systems may be evaluated with minimum effort, using data either already available or readily obtained (Petschel-Held *et al*., 1995).

Ecosystems have a high degree of aggregation, and consequently any management operation will affect the various components at different levels. One will have to select the level at which useful indicators can be derived to describe and evaluate changes in the system. Findings obtained at levels of high resolution (little aggregation) may not be applicable or scientifically relevant to decisions at higher levels of aggregation. Results of ecosystem research carried out at different levels within biological systems (Table 12.1) need to be synthesized for each level in terms of ecosystem responses based on the biochemical, physiological, population-dynamic, geological and pedological state of knowledge (Ulrich, 1987). Ulrich (1994) suggested methods to integrate the information obtained at different levels by using a hierarchical approach, considering the role of processes in forest ecosystem functioning. Such a treatment

Table 12.1. Research strategies and their outcomes at different levels within biological systems (Ulrich, 1987).

Biological system	Research strategy	Resulting knowledge on:
Ecosystem	Matter balance (deviation from the steady state)	Biologically caused changes of the soil, water quality and air quality (sustaining productivity and environment factors)
Population	Study of growth, yield and interactions between populations (e.g. competition, interactions between host and parasite)	Maximizing production of desirable biomass
Organism	Metabolism, physiology	Normal metabolic functions and deviations (disease)
Cell	Cell metabolism, biochemistry, molecular biology	Fundamental life processes and inheritance, normal cell metabolism and deviations (disease at cell level)

of ecosystem components would need the development and use of indicators at different levels. For each of the different levels, a hierarchical system can be developed to describe and evaluate changes in the components of forest ecosystems caused by forest management (Petschel-Held *et al.*, 1995).

The indicators used to describe and evaluate the proposed criteria can be grouped into various levels, depending upon their complexity. For example, there are basic indicators which are easy to obtain by analytical procedures. Others are highly complex, describing functioning at the top level of the hierarchy (system–normative indicators). This grouping of indicators is explained below, with examples:

1. *Analytical indicators* are those which describe the state and functioning of a component of the system. They are of little or no value for evaluation, but can provide quantitative differentiation between large and small, or positive and negative, changes. Simple analytical indicators of a system are measures which can be evaluated or grouped to provide a scale or a spectrum of possibilities of a change in the state or functioning of components of an ecosystem. Examples of such indicators are nitrogen concentrations in foliage or soil, or the concentration of a pollutant in rainfall.

2. *Compound indicators* are those which are formed by combining different analytically measured quantities of elements for various components of a system, thus allowing an additional description of changes in the system. One such example is the deposition rate of acids to a forest, which is the combined result of different inputs (bulk deposition, particle interception, gas interception). The rate of acid deposition has many implications for the health and functioning of ecosystem components such as vitality of fine roots, root uptake processes, mycorrhizal populations, litter quality and decomposition, and faunal activity. Another example of a compound indicator is the biological availability of nutrient or toxic substances in soils, for which a number of factors are to be considered such as the physico-chemical characteristics of the substance in question, the characteristics of the soils, and the characteristics and distribution of the organs responsible for uptake. The compound indicator of biological availability will provide information on the growth and vitality of system components depending on the direct physiological role of the element, and its interactions (and loading) with other system components. Compound indicators are usually obtained by combining a number of observed or measured quantities.

3. *System indicators* are those which cannot be measured or observed directly, but are derived from other system-based properties. They combine the analytical and compound indicators in a complex way to describe system characteristics such as complexity, diversity, stability, elasticity, resilience, linking and development potential. For instance, the significance of the impacts of a management practice on biodiversity cannot be determined by simply counting the number of organisms or species in an ecosystem, but would need consideration of the functional role of the impacts, such as effects on the food web, on the pattern of turnover of matter and energy within the ecosystem, and on species reproduction.

4. *Normative indicators* permit evaluation of high-level components such as ethic, social, economic or political factors. Normative indicators provide information about the quality of a system and its development for human needs. Natural sciences alone do not provide these indicators. Knowledge of causes and effects is essential to derive norms, but this in itself does not allow a decision on the appropriate action, which may be taken for social, economic and political reasons. The choice of action is influenced by the willingness of humans to take risks (Honnefelder, 1993). For example, a forest may have been severely damaged in its state and functioning but may fall in the expected norm (normative indicator) because of economic or social reasons.

As with analytical indicators, normative indicators can be obtained for different levels of aggregation and complexity in an ecosystem. Analytical indicators can be transformed into normative ones by developing suitable standards for their evaluation. For instance, setting a threshold value for any quantitative change may transform an analytical indicator into a normative one. By using this approach, the indicators which are required for multi-functional forest use, and that possess a normative character, can be obtained. They are applied by humans specifically to satisfy their demand for resource utilization and for sustainable management of forest ecosystems.

4 An extended critical loads concept and its application to define indicators for assessing change in forest ecosystems

A system–normative indicator of forest ecosystems can be best described by its attributes which measure change in the functioning of various components of an ecosystem. This can be done by employing an extended concept of critical loads and will be discussed by considering the capacity of the soil component to tolerate critical amounts of inputs (e.g. acid, nitrogen or other substances), critical level of forest operations (e.g. soil compaction or surface sealing) or critical amount of loss (e.g. soil or element losses by erosion). Changes induced by a critical amount of input or loss, or a critical level of forest operations, may result in critical states and functions in forest ecosystems. In order to develop this extended critical loads concept, it is useful to employ a compartment model to select and apply indicators at the ecosystem level. This approach has the advantage that state variables of ecosystem components can be clearly classified, and the associated ecological processes can be expressed as fluxes or transformation rates. Data of either a quantitative or qualitative nature can be expressed in a model structure or condensed into usable indicators.

The following compartments can be distinguished in a basic model of an ecosystem: (i) the atmosphere; (ii) the standing biomass; (iii) the soil; and (iv) the groundwater zone. The soil compartment includes the draining horizons down to the groundwater zone and can be further differentiated into the humus layer (compartment iiia) and mineral soil horizons (compartment iiib). The soil compartment adjoins the area permanently saturated with water (groundwater zone, compartment iv).

The concept presented here is based on comparing ecological limits with the extent of loading, and is an extension of the critical loads concept (Nielsson, 1986) which was developed in connection with air pollution and its effect on forests (Beese, 1992). This concept has so far been limited to fluxes of matter and has been applied to acidification (Acid Rain, 1995), nitrogen eutrophication (Acid Rain, 1992), ozone (Erisman *et al.*, 1998) and heavy metals (Paces *et al.*, 1998). The concept is based on fluxes in energy and matter extending beyond the boundaries of the system, resulting from an overloaded state of the ecosystem and possibly causing degradation of the system. An evaluation framework is needed to quantify man-induced changes and to assess their significance in terms of resource conservation, functioning of ecosystems and sustainable use of resources. Such a framework can be based on a quantification of loads with respect to breaking points in the system.

The types of loading given above are described below with examples to show their limitations and use in relation to the concept of multifunctional forest use (Beese, 1992):

4.1 Indicators for critical levels and critical loads

Concentrations of chemicals (e.g. O_3, SO_2, NH_3, NO_x) that exceed particular limits may result in direct damage to plant organs. These limits represent critical levels which should be avoided. Examples of critical loads for the deposition of acid and nitrogen are given in Tables 12.2 and 12.3. Other possible examples include heavy metals, organic substances or salts in forest soils.

The critical load of acid is related to the ability of soils to buffer the acid inputs to levels which present no ecological harm to plants and their functioning. At a soil pH below 4.2, the buffering capacity of soils may be very high because of the dissolution of aluminium hydroxides and clay minerals, but the release of hydrolysing cations (Al^{3+}, $Fe^{2+/3+}$) during buffering of protons will have toxic effects on plants and soil organisms. Thus indicators for critical acid loads should include soil buffering capacity and the rate of release and the concentrations of toxic cations (Al^{3+}, $Fe^{2+/3+}$). The buffering capacity for protons will be affected by the mineral composition of the parent material (weathering of minerals) and will determine the critical acid load (Table 12.2).

The critical load for nitrogen will depend upon the state of the particular ecosystem and the extent of leakage of nitrogen. High loads of N can cause degradation of neighbouring systems (e.g. groundwater). The concept of critical nitrogen loads is still controversial. Some estimates of critical loads in various ecosystems and their reliability are given in Table 12.3.

4.2 Indicators for critical levels of forest operations

The impacts of forest operations on a site can be considered as being physical, chemical or biological in nature, and can be examined in terms of critical loads by assessing the intensity of changes in the structure and functioning of forest ecosystems.

Table 12.2. Critical loads of acid in soil in relation to the mineral composition and parent rock material (Acid Rain, 1995).

Minerals which determine the buffer rate	Parent rock material	Critical acid input ($kmol_c$ H^+ km^{-2} $year^{-1}$)
Quartz/K-feldspar	Granite, quartzite	< 20
Muscovite, plagioclases, biotite (< 5%)	Granite, gneiss	20–50
Biotite, amphiboles (< 5%)	Granodiorite, greywacke, slate, gabbro	50–100
Pyroxenes, epidote, olivine (< 5%)	Gabbro, basalt	100–200
Carbonates	Limestone	> 200

An example of a physical change is soil compaction and deformation during harvesting operations. Among other things, soil compaction affects the penetrability of soil to roots, and it is important to define, for different tree species, the critical (limiting) values of root penetrability which should not be exceeded. Simple indicators can be employed to relate soil compaction to root penetrability. For example, Rab (1999) used changes in bulk density, aeration porosity and the area affected by subsoil disturbance as useful indicators. Soil compaction may also change water permeability so that heavy rain results in surface runoff and erosion. A measure to evaluate this change could either be the exceeding of a critical value of soil loss, or changes in particle size distribution and organic matter content (Sands *et al.*, 1979; Rab, 1996). Rab (1999) suggested that a simple indicator for the area of forest land with significant soil erosion could be the sum of area affected by access roads, landings, snig tracks, firebreaks and subsoil-disturbed harvest areas. However, it would also be important to take the inherent erodability of the soil into account.

An example of a biological change is the temporary removal of vegetation during clear-felling which may induce critical levels (loads) of soil change, resulting in erosion by water or wind, enhanced slope movement or nutrient losses (Bormann *et al.*, 1974; Ludwig *et al.*, 1997). Nutrient export in biomass may be critical if the nutrient resources available in the soil, or natural inputs of nutrients, are too small to rebuild a healthy and productive forest stand. The introduction of an exotic tree species is another example of biological change that may impact on the site, especially if the species is not adapted to site conditions. Introduction of a new set of species can change biodiversity

Table 12.3. Critical loads for nitrogen in various ecosystems (Acid Rain, 1992).

Ecosystem	Critical nitrogen load (kg N ha^{-1} $year^{-1}$)	Reliability of estimate of load[a]
Shallow soft-water bodies	5–10	xxx
Mesotrophic fens	20–35	xx
Ombrotrophic bogs	5–10	x
Calcareous species-rich grassland	14–25	xxx
Neutral-acid species-rich grassland	20–30	xx
Montane-subalpine grassland	10–15	x
Lowland dry-heathland	15–20	xxx
Lowland wet-heathland	17–22	xxx
Species-rich lowland heaths/acid grassland	7–20	x
Arctic and alpine heaths	5–15	x
Acidic coniferous forest	10–20	xxx
Acidic deciduous forest	< 15–20	xx
Calcareous mixed-species forests	15–20	xx

[a]xxx = reliable; xx = quite reliable; x = best guess.

and affect soil processes, productivity, resource utilization and ecosystem functioning. It may not be possible to assess critical loads for all the factors involved, but some indicators for biodiversity and soil change are required. Some simple indicators for biological changes have already been suggested under Montreal Process Criterion 3: (3a) the area and percentage of forest affected by processes or agents beyond the range of historic variation (e.g. by insects, disease, competition from exotic species, fire, storm, land clearance, permanent flooding, salinization and domestic animals), and (3c) the area and percentage of forest land with diminished biological components indicative of changes in fundamental ecological processes (e.g. soil, nutrient cycling, seed dispersion, pollination) and/or ecological continuity (functionally important species such as nematodes, arboreal epiphytes, beetles, fungi and wasps).

4.3 Indicators for critical states and functions

Critical states and functions in forest ecosystems arise when the physical and chemical states change in the long term, or the biotic states (plant, animal and microorganism communities) change in such a way that productivity, stability and biological diversity is adversely affected. These changes can be caused by inappropriate material inputs and losses, mechanical disturbance or changes in the biological environment.

Structural indicators of a possible critical state or function are shear resistance, soil compactness, pore distribution and form, humus content, degree of base (Na, K, Mg, Ca) saturation of exchangeable sites, Ca/Al ratio in the soil solution, the composition and mass of the biological community, and the amount and concentration of nutrients and toxic elements. The nominal values of base saturation required for different tree species (Ulrich, 1995) are shown as an example in Table 12.4. If the base saturation falls below such values a critical state may be induced.

Table 12.4. Nominal minimum values for the degree of base (Na, K, Mg, Ca) cation saturation for different tree species (Ulrich, 1995).

Species (common name)	Nominal value (%)
Acer campestre (field maple)	90
Ulmus glabra (Wych elm), *Fraxinus* spp. (ash), *Tilia cordata* (small-leaved lime)	70
Acer platanoides (Norway maple), *Prunus* spp. (cherry)	60
Acer pseudoplatanus (sycamore), *Carpinus betulus* (hornbeam)	50
Fagus sylvatica (beech), *Quercus* spp. (oak), *Picea* spp. (spruce), *Abies* spp. (fir), *Pseudotsuga menziesii* (Douglas fir), *Pinus* spp. (pine)	> 30

Other examples of indicators of critical states are given by Meiwes *et al.* (1984), Ulrich *et al.* (1984) and Cronan and Grigal (1995) who suggested that base saturation and molar Ca/Al ratio in soil solutions, fine roots and current foliage be determined in order to estimate the risk of an Al stress in forest ecosystems (Table 12.5).

Critical functions could be proton buffer rates, weathering rates of soil minerals, rates of nitrogen mineralization or soil organic matter decomposition, transport of water and gas through soil, or the growth of plants and soil organisms.

In general, indicators suitable for defining critical soil internal states are still not developed. There are some guidelines for toxic concentrations of some elements or element ratios, but most of these values refer to potential effects on organisms (e.g. on human health via the food chain, or on tree survival and growth). Indicators pointing to impaired functioning of system processes are less known. However, for defining site conditions, indicator plants and plant communities have sometimes been used, but no reliable criteria exist for animal and microbial communities. Further, there is little information available on the sensitivity and speed of response of currently used plant indicators of change in site characteristics.

4.4 Indicators for critical amounts of loss

The critical amount of loss of matter or organisms from a forest ecosystem refers to the effect of the loss on the functioning of the source system or the effect of increased inputs on the vitality and functioning of neighbouring systems. The resulting loads in neighbouring systems need to be evaluated for individual components (humans, animals, groundwater, atmosphere, etc.). Critical amount of loss may restrict the way forest resources are used because of the danger of degradation of the neighbouring system (e.g. an increase in the

Table 12.5. Multiple assessment tools for determining whether an ecosystem has a high probability of suffering Al stress (Cronan and Grigal, 1995).

Measurement endpoint	Threshold
Soil base saturation	< 15%
Soil solution Ca^{2+}/Al_i molar ratio	1.0 (50% risk)
(Ratio of Ca^{2+} to inorganic charged Al)	0.5 (75% risk)
	0.2 (95–100% risk)
Fine root Ca/Al molar ratio	0.2 (50% risk)
	0.1 (80% risk)
Current foliage Ca/Al molar ratio	12.5 (50% risk)
	6.2 (75% risk)

nitrate loads of groundwater if used for drinking purposes will restrict any operation in the catchment areas which may affect that water). There are examples of the critical amount of soil loss due to erosion, or nutrient loss due to leaching, which have affected the functioning and health of forest ecosystems.

Already some practical examples of indicators for critical losses, especially where human health is affected directly, are available in guidelines. Internationally binding guidelines for acceptable levels of certain substances in drinking water and food have been established by the European Union and World Health Organization.

5 Conclusions

The extended critical loads concept presented here considers significant changes in the functioning and use of forests. It consists of critical amounts of material inputs (loads), critical levels of forest operations, critical states and functions, and critical amounts of loss from terrestrial ecosystems. This concept includes site properties as a factor affecting impacts within landscapes, and considers time as an important variable. This concept provides a useful conceptual framework for developing and evaluating indicators that reflect the functioning and vitality of forest ecosystems.

In future, we need to strive for a greater integration of habitat use, cultural and social values, resource utilization and regulatory mechanisms for multifunctional forest use. This can only be achieved through cooperative interdisciplinary efforts.

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Impacts of Environmental Stress on Forest Health: the Need for More Accurate Indicators

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Environmental stresses, including both natural and anthropogenic phenomena, can pose significant risks to sustainable forest management. Natural environmental stresses are not generally considered to be problematic over timescales of hundreds to thousands of years. However, forests are normally managed over shorter time periods, and losses through, for example, fire may significantly affect the economic sustainability of a forest. Environmental stresses induced by anthropogenic processes also represent a threat to sustainability and they may require action at regional, national or international level. Some of the existing indicators of environmental stress are still rather limited and require further development if their full potential is to be realized. New indicators proposed here, some of which are already in use in some countries, are: area of forest exceeding critical loads and levels (or other air pollution standards), proportion of forests with soils saturated by nitrogen or with negative balances for critical elements, area of forest adversely impacted by exotic pests and pathogens, proportion of cut timber with serious quality problems, proportion of timber harvest classed as salvage felling, proportion of forest managed through natural regeneration, proportion of forest planted with genetically 'improved' trees, percentage of standing dead trees, and a measure of genetic diversity to monitor the adaptability of the forest to future environmental changes. Several of these require further developmental work before they can be fully implemented.

1 Introduction

Although the maintenance of forest health is frequently cited as an important criterion for sustainable forest management (SFM), a clear, concise, and reproducible definition of forest health has not been developed. Several definitions have been proposed, but these differ considerably between interest groups. For example, a production forester working with plantations may consider a healthy forest to be one that is free of disease and producing the expected yield of timber. Conversely, an ecologist is likely to view a healthy forest as one in which all expected ecological processes are occurring. A political ecologist will also include social and economic processes in the definition. As a result, it is not possible to provide a definition of forest health that is applicable to all parties or forests. Instead, we suggest that forest health be assessed in relation to the expectations for a particular forest (Society of American Foresters, SAF, 1997) and with the knowledge that these expectations may change over time. If a forest meets the expectations of its stakeholders, and its condition is considered to be good (which is also a value judgement), then it might be considered to be healthy, at least by the stakeholders. This notion departs from more traditional viewpoints that consider forest health within the context of the incidence and severity of insects and diseases, but seems to be consistent with current ideas on ecosystem management (cf. Salwasser, 1998). Sometimes the health of the ecosystem components (e.g. the soil) is of major interest as well as the health of the 'forest'. This is the case with afforestation of degraded land (Fig. 13.1).

Environmental stresses are determinants of forest condition (Bazzaz, 1996). Although disturbances are natural in forests, both natural disturbances and those induced directly or indirectly by human activities can temporarily or permanently reduce the extent to which a forest fulfils the expectations held for it. For example, forest reserves are frequently established to protect a particular species or ecosystem type. A natural disturbance, such as a hurricane, may reduce the conservation value of the reserve by altering the conditions required by the habitat type or species. Such situations arise with increasing frequency as the relative importance of reserves increases as a result of progressive forest losses and the occurrence of anthropogenic disturbances such as fire increases due to increasing populations and the encroachment of urban areas into forests.

Environmental stresses are addressed in many documents that discuss criteria and indicators (C&I) for SFM (Table 13.1). In several cases, the proposed indicators are not easily related to the criteria under which they are listed. For example, under the Helsinki Process, the deposition of air pollutants is given as an indicator. However, the absolute amount of pollutant deposition at a site may bear little relation to its potential to endanger the long-term forest sustainability because the buffering capacities of ecosystems differ. Consequently, there is a need to examine environmental stress factors that

Fig. 13.1. In many parts of the world, afforestation is being undertaken to help arrest or reverse degradation of rural landscapes. In this example, a break of slope planting of eucalypts is being used to intercept water flowing from upland (recharge) areas to lower parts of the landscape where it contributes to raising of water tables and the development of dryland salinity. Measures of the multiple benefits (creation of habitat, enhanced water use, carbon sink creation, economic return from forest products) are needed to help evaluate the effectiveness of such activities.

affect forests, to derive appropriate indicators for each stress and to relate these indicators to established criteria.

2 Important stress factors

Potential environmental stressors include natural phenomena such as climate and anthropogenic phenomena such as air pollution. Most natural stressors can also be modified by human activities, as for instance with anthropogenic impacts on climate. Examples of natural and anthropogenic stressors are given in Table 13.2. In interpreting these, it must be realized that the distinction between anthropogenic and natural causes is often unclear, that there may be interactions between different stress types, and that particular stresses can affect forests differently, as illustrated by the decline spiral of Manion (1991) in Fig. 13.2.

Table 13.1. Direct or indirect indicators of environmental stresses listed in the various C&I initiatives.

Helsinki

2. Maintenance of forest ecosystem health and vitality
   - 2.1 Total amount of and changes over the past 5 years in depositions of air pollutants (assessed in permanent plots)
   - 2.2 Changes in serious defoliation of forests using the UN/ECE and European Union defoliation classification (classes 2, 3, and 4) over the past 5 years
   - 2.3 Serious damage caused by biotic or abiotic agents
     - a. severe damage caused by insects and diseases with a measurement of seriousness of the damage as a function of mortality or loss of growth
     - b. annual area of burnt forest and other wooded land
     - c. annual area affected by storm damage and volume harvested from these areas
     - d. proportion of regeneration area seriously damaged by game and other animals or by grazing
   - 2.4 Changes in nutrient balance and acidity over the past 10 years; level of saturation of exchange capacity on the plots of the European network or of an equivalent national network

Montreal

3. Maintenance of forest ecosystem health and vitality
   - a. Area and percentage of forest type affected by processes or agents beyond the range of historic variation, e.g. by insects, disease, exotic competition, fire, storm, land clearance, permanent flooding, salinization and domestic animals
   - b. Area of forest subjected to levels of specific pollutants (e.g. sulphates, nitrate, ozone) or ultraviolet B that may cause negative impacts on the forest ecosystem
4. Conservation and maintenance of soil and water resources
   - a. Area of land with significant soil erosion
   - d. Area and percentage of forest land with significantly diminished soil organic matter and/or changes in other soil properties
   - g. Percentage of water bodies in forest areas (e.g. stream kilometres, lake hectares) with significant variation from the historic range of variability in pH, dissolved oxygen, levels of chemicals (electrical conductivity), sedimentation or temperature change
   - h. Area and percentage of forests experiencing an accumulation of persistent toxic substances

Amazon Cooperation Treaty

4. Conservation of forest cover and regional biological diversity
   - d. Area and percentage of forest affected by processes or other agents (insect attack, disease, fire, flooding, etc.)
   - g. Area and percentage of forest lands with fundamental ecological changes
5. Conservation and integrated management of water and soil resources
   - c. Percentage of forest flooded in relation to the historic range of variation and maintenance of the relationship between the forest and hydrobiological resources

Table 13.1. cont'd.

10. Conservation of forest ecosystems
c. Area and percentage of forest affected by processes or other agents (insect attack, disease, fire, flooding, etc.) and by human actions
FAO/UNEP Far East Forests
3. Health, vitality and integrity
1. Areas and percentage of forest (plantations/natural forests) affected by: natural fires; storms; insects and diseases; drought; wild animals (game)

FAO, Food and Agriculture Organization of the United Nations; UNEP, UN Environmental Programme.

3 Thresholds for stresses

In relation to SFM, a stressor is only important if it threatens the values that are perceived to exist for a forest. Many stressors occur at background levels with few or no long-term effects. For example, tree foliage is damaged each year by herbivores, but this feeding does not *necessarily* have an impact on the long-term health of either the trees or the forest. Every organism within a forest ecosystem has an environmental tolerance. If the tolerance limits are not exceeded, then the organism will be able to survive and reproduce. The tolerance limits for an individual may be increased by interactions with other individuals within the ecosystem. An example of this is the increased tolerance of a group of trees to wind over that of a single tree. However, there are many subtle interactions within ecosystems (only some of which are well-documented) that may enable organisms to cope with stresses that as an individual they would not be able to tolerate.

Historically, environmental stressors such as storms and fire were seen as threats to forestry; today, in some areas, they are considered to be an integral part of the natural disturbance regime. However, the presence of such disturbances in forests which have wood production as a primary aim is usually unwelcome. The attitude of foresters to such disturbances depends on the nature of the forests they manage. For example, in the semi-natural forests of western and north-eastern North America, disturbances are seen as a part of the system and, to a certain extent, tolerated. In western Europe, with its much more intensively managed forests and the close relationships between forests and human settlements, the general acceptance of such disturbances is much lower.

3.1 Natural disturbance regimes

Considerable debate surrounds the nature of natural disturbance regimes. This concept is scale-dependent in both space and time. Consequently, in some

Table 13.2. Stresses in forests that could jeopardize their long-term sustainability.

Natural stresses	
Climate	Drought Extreme precipitation (including snowfall, hail, rain, etc.) Lightning and associated fires Extreme temperatures (including frost, heat, etc.) Wind: storms, hurricanes, tornadoes, etc. Ice storms
Geomorphological events	Floods, erosion, landslides, avalanches, volcanic activity, etc.
Soil	Nutrient deficiencies Waterlogging Acidification Salinization
Stand dynamics	Successional processes
Pathogens, parasites and other organisms causing damage to forests	Insects, fungi, viruses, nematodes, mammals, etc.
Anthropogenic stresses	
Air pollution	Gaseous pollution: SO_2, NO_x, CO_2, O_3, HF, PAN, etc., wet and dry deposition of heavy metals, sulphate, nitrate, ammonium, etc.
Climate change	Changes in the frequency or magnitude of natural climatic stresses, elevated temperatures
Global processes	Increases in UV-B radiation
Soils	Eutrophication Acidification Heavy metals Erosion Nutrient losses
Forest management	Tree-cutting Over-harvesting Soil compaction Litter removal Damage to residual trees during harvesting Poor site-species matching Introduced pathogens Fire

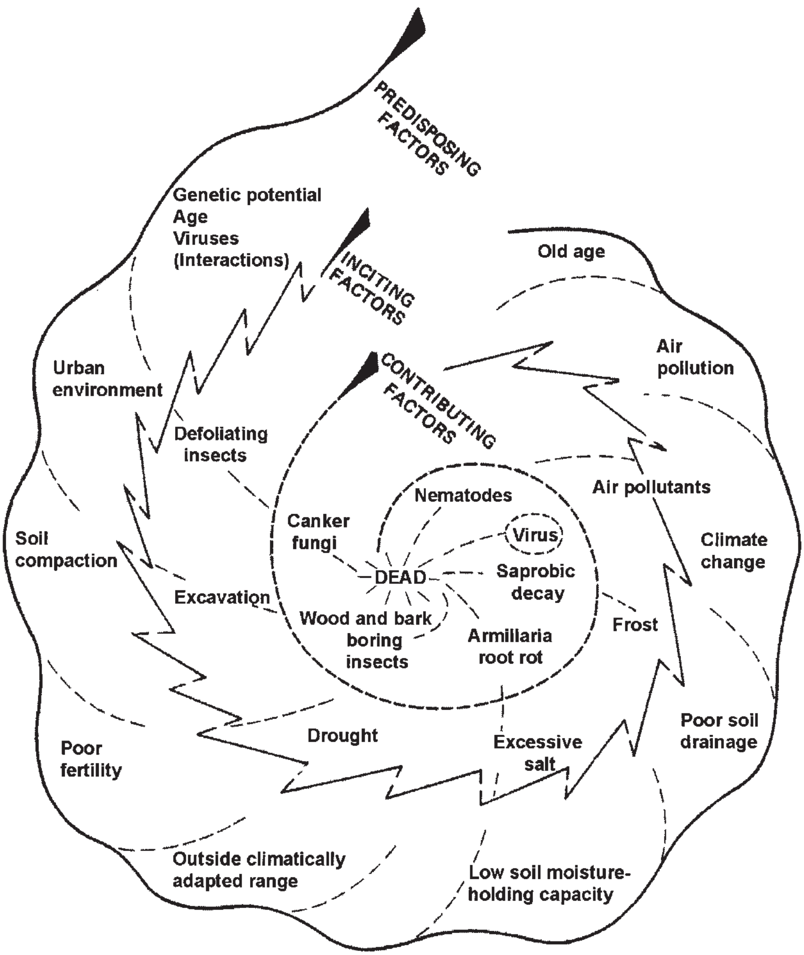

Fig. 13.2. Decline disease spiral. The figure suggests that only an inward spiral can occur, but in some cases appropriate remedial action can enable a tree to recover. Redrawn from Manion (1991).

documents (e.g. Montreal Process 3a, 3g, Amazon Cooperation Treaty 5c) the term 'natural range of variation' has been adopted. This term is often difficult to apply, as the historic range of variation is frequently unknown. The difficulty is well illustrated by studies of climate variability reconstructed from tree rings. For example, work by Hughes and Graumlich (1996) and Hughes and Funkhouser (1998) in the Great Basin area of western North America indicates that moisture deficit extremes during the last 1000 years have been more intense than those experienced in the 20th century. Consequently, even if detailed information is available for the last 100 years, such information does not necessarily provide a good indication of the historic range of variability, nor does its availability mean that we would necessarily wish to mimic it through management.

All disturbances have a magnitude–frequency distribution. Small disturbances tend to be more common than larger ones, and extreme events are, by definition, rare. This generalization is site-specific. For example, the magnitude of forest fires varies regionally, as does their recurrence interval (cf. Oliver and Larson, 1996). Some disturbances can be very large, such as those individual fires in boreal forests which have burnt over 1 million ha. When considered over a sufficiently large area and over a sufficiently long time-scale, such large-scale disturbances might be acceptable to forest management. However, today, it seems unrealistic to view such events as a requirement for SFM, which is necessarily restricted to smaller areas and shorter time-scales. Consequently, the degree of acceptance is likely to be dependent on the relationship between the magnitude of the disturbance and the size of the forest management unit (FMU).

Difficulties associated with the identification of the natural range of variability are confounded by anthropogenic effects, as many processes in all managed forests have been influenced by man. This influence is particularly apparent in Europe and eastern North America, but many other areas have been affected as well. Today, most ecologists see forests as a mosaic of different stands with different developmental stages (cf. Oliver and Larson, 1996; Kimmins, 1997). Sizes of individual patches within the mosaic can vary, depending on the type and extent of the disturbances affecting them. The concept of a time-space mosaic is important when looking at SFM practices, as it adds another dimension to considerations of the roles of disturbances. For example, catastrophic storms occur naturally in some forest types and, while the short-term production of timber may be adversely affected, the system may be seen as coping with the storms if a sufficiently long time-scale is adopted. As natural forests are dynamic, with major ecosystem changes possible, forest management should not necessarily be restricted to the preservation of a forest landscape that represents a theoretical ideal.

3.2 Loss of genetic diversity

Environmental stresses can affect the ability of trees to survive (viability selection) and to reproduce (fertility selection). However, whether germplasm is lost and genetic diversity is impacted by environmental stresses is still under debate (Pitelka, 1988). Karnosky (1981) and Berrang *et al.* (1991) have presented evidence of tree population changes induced by air pollution. Indicators of this type of change are subtle, difficult to detect and generally involve isozyme analysis (Scholz and Bergman, 1984; Müller-Starck, 1985) or molecular methods such as random amplified polymorphic DNA (Bucci *et al.*, 1997) or restriction fragment length polymorphism (Neale and Williams, 1991). However, with the rapid developments that have been made recently in population genetics, such information is becoming increasingly available.

The importance of managing genetic structure as a component of sustainable forestry for stressed forests has been highlighted by Karnosky and Scholz (1996). Another important point related to genetic diversity is the extension of many species outside their natural range as a result of forestry activities. Little is known about how the reactions of these introduced populations to stress may differ from the reactions of natural populations of the same species.

3.3 Irreversible changes in forest ecosystems

A principal aim of SFM is to reduce the likelihood of irreversible changes that could affect forest values. Decline and mortality of all trees in a forest would clearly represent such a change, although it might be only a short-term phenomenon. Assessing the likelihood of such changes would require detailed ecological risk analyses. The extent to which these analyses have been conducted is extremely limited as they are expensive and difficult to design and implement. Even in the USA, the Environmental Protection Agency has concentrated its risk analyses on the potential effects of air pollution on the timber production of an economically important species (loblolly pine, *Pinus taeda*) whilst paying much less attention to the potential impacts of pollutants such as ozone on forest ecosystems as a whole.

Insufficient attention has been given to the manner in which forest ecosystems can develop and change through time. The traditional view that forests develop via successional processes towards a single climax state (Clements, 1928) is now considered to be over-simplified. Instead, there appear to be multiple end-points, with individual stands within a forest progressing towards one of these until a disturbance starts the process again (with the possibility that the stand then starts developing towards another end-point) (Miles, 1987; Oliver and Larson, 1996; Peterken, 1996; Richards, 1996). This dynamic pattern must be placed within the context of a changing environment: the chemical and physical environment of forests throughout the world today is unique, and will continue to change in the future (Mohren *et al.*, 1997).

4 Stresses within the context of C&I to improve forest management

The indicators in Table 13.1 need to be examined in detail to assess their value in detecting change over time for each criterion. They can be divided as follows:

Stress indicators
- Air pollutant deposition (Helsinki 2.1)
- Air pollutant exposures (Montreal 3b)
- Toxic substance accumulation (Montreal 4h)

Response indicators

- Defoliation of trees (Helsinki 2.2)
- Damage caused by biotic or abiotic agents (Helsinki 2.3, Montreal 3a, Amazon 4d, 5c, 10c, Far East 3.1)
- Changes in soil chemistry (Helsinki 2.4, Montreal 4d)
- Area with soil erosion (Montreal 4a)
- Water chemistry (Montreal 4g)
- Areas showing ecological changes (Amazon 4g)

Information on the deposition of air pollutants will not provide sufficient information about the environmental risks they present to forests. Instead, deposition needs to be related to the sensitivity of forest ecosystems to such processes. Sensitivity analyses have been undertaken in Europe through the critical loads approach. A critical load is defined as 'the highest deposition of a compound that will not cause chemical changes leading to harmful effects on ecosystem structure and function' (Nilsson and Grennfelt, 1988), although what constitute 'harmful effects on ecosystem structure and function' are not defined. Critical loads have been calculated for many different types of forest soils (e.g. Rihm, 1994) and have been mapped at a European scale. From these efforts, maps of critical load exceedances have been derived. Although there are still many uncertainties with such figures, they provide a more meaningful assessment of the likely effects of the deposition of pollutants than do figures for pollutant deposition alone. They also enable priorities to be set for pollutant emission reductions.

The same principle has been applied to gaseous pollutant concentrations, although the term given is critical level. The concept is essentially the same as the secondary standard used in the USA, although the latter has not been specifically derived for forests. There is considerable disagreement over the choice of exposure index for gaseous pollutants, exemplified by the differences between European and North American methods used for characterizing ozone exposure. In reality, exposure to a gaseous pollutant is not the same as the dose received, with the latter being dependent on stomatal uptake. Currently, attempts are being made to improve estimates of the doses received by plants in forests, and the use of more traditional measures, such as mean annual concentrations, should be treated with considerable caution.

Accumulation of toxic substances provides a useful indicator for SFM. Its main limitation is our lack of knowledge about the toxicity of many substances that currently are being released into the environment. In addition, the substances may react with each other or with naturally occurring substances, resulting in the formation of new compounds about which even less is known. The risks presented by such accumulations can be assessed through bio-monitoring (e.g. van Straalen and Lokke, 1997; Jørgensen and Halling-Sørensen, 1998).

With all the different pollutant concepts, there is a need for further research on the tolerance of forests to pollutant stress. The AOT40[1] standard allows for a 10% reduction in growth before the critical threshold is crossed. However, in many areas where this AOT40 standard is exceeded, available growth data do not suggest a growth reduction. These growth data suggest that either the forest has buffering systems that influence its response to ozone, or that other factors more than compensate for any growth losses attributable to ozone.

In contrast to indicators such as critical load exceedance, the defoliation of trees represents a response indicator. Although widely used in Europe, the value of defoliation as an indicator of forest health has been questioned (e.g. Innes, 1988; Skelly and Innes, 1994). It fails to meet the requirements of most environmental indicators in that its relevance to the criterion of forest health is uncertain and the measurements are not reproducible. One of its greatest drawbacks is that it is non-specific. Defoliation is assessed in relation to crown transparency, so it is a combined index of foliage that has been shed, foliage that is smaller than usual, and foliage that has never developed. The extent of the natural variation in this index is unknown, as is the background level that should be taken as normal. If the natural levels of defoliation could be established, then its value as a general indicator would be significantly enhanced. Similarly, if more quantitative assessments of defoliation were available that could be used to calibrate the visual assessments, then the reliability of the indicator would be improved.

The indicator that is the most cited in the C&I documents is the extent of damage by biotic and abiotic factors. As with defoliation, its interpretation is very dependent on knowing the background levels of damage. The interactions with anthropogenic factors such as air pollution and climate change remain largely speculative as the majority of work to date has been of an experimental nature conducted under controlled conditions, which has little relevance to the real situation in forests (Docherty *et al.*, 1997). In some cases, an epidemiological approach may help to identify possible interactions, but these then need to be confirmed through carefully designed field experiments. While it is clear that introduced pathogens, such as chestnut blight, and damaging agents, such as the balsam woolly adelgid or pinewood nematode, present a major threat to some forests, natural population fluctuations of indigenous species can also provide a temporary threat to some forest values.

Many forest insects show cyclic population fluctuations or episodic outbreaks, and the extent of damage is therefore also cyclic or episodic. Consequently, any attempt to relate such fluctuations to sustainable management must take these dynamics into account. Finally, it is unclear to what extent some forest management policies are in direct contradiction. For example, while wildlife habitat trees and dead wood are being promoted in some policy documents, their roles as potential sanitary hazards (e.g. as sources of further infection or infestation) still needs to be evaluated.

Changes in soil chemistry appear to be a fairly good indicator of environmental stress, provided that they are correctly interpreted. Changes in soil chemistry are a natural consequence of forest development. The challenge will be to separate such changes from changes occurring as a result of problems such as acidic deposition. Such distinctions are possible, as shown by the work of Tamm and Hallbäcken (1988). Much emphasis has been given to changes in soil acidity, but this may to some extent be misdirected. Other measures, such as base saturation, cation exchange capacity and the available concentrations of specific elements may be more useful than soil pH as indicators of forest health and indeed are being applied in some cases.

Soil erosion is also a useful indicator. It should be interpreted broadly, to include all soil losses from catchment areas, particularly those resulting from mass wasting. It can be measured by a variety of means, but demonstrating a link to forest management practices may be more difficult. Some soil erosion is to be expected in all catchments: it is part of the natural process of denudation. As with many other indicators, it is the difference between background levels of erosion and accelerated rates resulting from anthropogenic activities that is important. Simply measuring changes in rates will be inadequate, as natural rates change over time, depending on natural fluctuations in the climate–soil–vegetation system within a catchment. However, there is a threshold for soil erosion that can often be identified without too much difficulty because the visual effects are so dramatic. Losses in soil cover is one example, evidence of siltation in river beds is another.

Change in water chemistry is also a reasonably clear indicator although, as with all indicators, it must be supported by good science. Clear cause–effect pathways must be demonstrated before the links between forest practices and water quality can be developed. In addition, the role of forest management practices must be placed within the context of the whole ecosystem. For example, in the 1980s there was considerable debate over the role of afforestation in the acidification of freshwater streams in upland Britain. The argument was put forward that forests encouraged more occult deposition of acidic pollutants than grasslands, and that afforestation should therefore be halted. In such a case, forest management was blamed for a problem that should really have been controlled at source, namely the release of acidic pollutants.[2]

The use of areas with ecological changes as an indicator of environmental stress is probably of limited value as so many different factors could cause such changes. In practice, the use of ecological changes as indicators of sustainable forestry needs to be very carefully defined. For example, changes in forest composition as a result of successional processes are clearly not grounds to doubt the sustainability of forestry in an area.

5 Revision of indicators

Any revision of indicators must be based first on a full acceptance of the criteria that they are supposed to support. Of the six criteria relevant to external environmental stresses given in Table 13.1, the criterion most in need of good indicators is the maintenance of forest ecosystem health and vitality, as three of the six deal with this theme. This concentration on forest ecosystem health and vitality presents a problem, as the criteria discussed in this chapter deal primarily with national-level issues, whereas forest ecosystem health and vitality is primarily an attribute of each individual system. As indicated earlier, it is unlikely that a particular ecosystem will be able to fulfil all the expectations that might be placed on it. Consequently, to make generalizations of the type used in C&I documents, it is necessary to increase the scale of assessment beyond that of the individual FMU. Scale was explicitly discussed in the first documents associated with the Montreal Process, but was given less emphasis in later documents. Moving from the stand upwards, the terms landscape and region are commonly used. Both these terms are difficult to define, particularly as regions may be larger than nations in some areas (e.g. Liechtenstein). However such difficulties should not prevent the specification of suitable indicators.

At the scale of the landscape, a mosaic of different ecosystem types can be envisaged. No two ecosystems would be identical, and each would have its own management prescription based on the local conditions, requirements and objectives. In large areas of publicly owned land, such as the National Forests of the USA or Crown Lands of Canada, the management of the mosaic could occur at a landscape scale to ensure that different ecosystem states were maintained. Similar management strategies could be adopted for large blocks of private land. Such a concept is much more difficult to apply in areas such as Europe, where small-scale land holdings are the norm. Any planning at the landscape level will involve conflict, as it implies strict control of the actions of individual forest owners in relation to the activities of others.

Taking into account these difficulties, potential indicators of relevance to forest ecosystem health and vitality include:

- suitable genetic diversity to ensure adaptability of the forest to future environmental changes;
- area of forest in exceedance of critical loads and levels (or other air pollution standards);
- proportion of forests with soils saturated by nitrogen;
- proportion of forest with negative balances for critical elements;
- area of forest adversely impacted by exotic pests and pathogens;
- proportion of cut timber with serious quality problems (e.g. heart-rot);
- proportion of timber harvest classed as salvage felling;
- proportion of forest managed by natural regeneration;
- proportion of forest planted with genetically ‘improved’ trees; and
- percentage of standing dead trees.

Each of these requires careful definition to ensure standardization in national assessments. For example, the indicator ‘area of forest adversely impacted by exotic pests and pathogens’ requires a definition for adverse impact, something that is outside the scope of this chapter. The applicability of each of these indicators is likely to vary between different forest types, and this variability needs to be taken into account when making any final recommendations.

The ability of individual managers to assess these proposed indicators will vary. Essentially, the first four deal with large-scale issues that are probably best dealt with by specialized institutions. The remainder consist of data that could be collected by forest managers fairly easily as a part of normal forestry operations.

6 Conclusions

There are clearly a number of problems with some of the indicators being proposed under the different C&I initiatives. In terms of environmental stresses, it is important to recognize that indicators are scale dependent, and that those that have been developed so far mostly concern national or regional criteria rather than criteria for the management of individual FMU. In addition, some indicators are independent of forest management policies. For example, a forest may be adversely affected by air pollution from industrial sources that have no connection to the forest. Trans-boundary air pollution is particularly difficult to deal with, and innovative methods are required to handle such cases. Any such procedure would clearly need to be based on a transparent and rigorously scientific procedure.

Scale is a critical issue that determines the acceptability of a particular disturbance. The magnitude and frequency of a disturbance in relation to the size of the management unit and the timescales over which it is managed will determine whether the disturbance threatens the sustainability of the management operation. In some cases, the forest management practices may have to be adapted to take this into account – as in Scotland where some Sitka spruce (*Picea sitchensis*) are felled prematurely to avoid wind damage.

There is a fundamental issue that still needs to be resolved. Are sustainable forests and SFM the same thing? As indicated in the introduction, sustainable forests can be defined as forests that meet all the expectations that are held for them. SFM involves actions by man that ensure that these expectations are attained. Over long timescales, all natural forests are probably ecologically sustainable, but they may not meet other expectations, such as timber yield. Consequently, different objectives need to be set for different forests. At the scale of the FMU, the different objectives make it impossible to draw up a set of indicators that is applicable to all forests. However, it would be possible to list indicators for particular sets of forest management objectives, such as nature conservation, timber yield or recreation. The extent to which multiple objectives can be achieved also needs to be examined. For example, to what extent can plantation forests established to supply timber satisfy the objectives of nature conservation? While much progress has been made towards answering such questions, there are still many areas of uncertainty that need to be resolved before fully operational sets of C&I can be brought into practice.

Notes

1 The AOT40 standard for European forests is a cumulative (6-month) measure of ozone exposure. All daylight (> 50 W m^{-2}) hours with an ozone concentration of ≥ 40 ppb are summed, with for example a concentration of 60 ppb for 1 h counting as 20 ppbh. The critical level is currently set at 10,000 ppbh, normally expressed as 10 ppmh.

2 In such a situation, economics can have a major impact on management strategies. It was much cheaper to stop afforestation than to control the pollutant emissions, and the end-effect (reduced freshwater acidification) was the same.

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14

Guiding Concepts for the Application of Indicators to Interpret Change in Soil Properties and Processes in Forests

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Protecting soil fertility is critical to the maintenance of many forest values that underpin sustainable forest management. In theory, indicators of soil fertility can be identified, monitored and evaluated to provide a basis for improved soil management. Only limited progress has been made, however, in translating this theory into practice. Practical progress will require greater consideration of indicator definition, monitoring and evaluation as linked processes.

The complexity of the soil system and the large spatial and temporal variation that occurs in important soil processes make it difficult to identify a small number of generic and relatively simple indicators of soil fertility. Soil indicators must reflect important soil functional processes, but these often do not show simple or direct relationships with forest ecosystem-level responses (e.g. species composition or growth). However, it is broadly accepted that measures of soil organic matter (SOM), soil acidity and base status, soil density and erodibility are relevant.

A set of field measures for describing soil physical and chemical change, soil pollution and erosion risk are proposed and discussed. A framework for operational monitoring of soil change that addresses spatial scale and temporal considerations is presented. This is illustrated for two native *Eucalyptus* forest regions of Victoria, Australia, that are intensively harvested and regenerated by broadcast slash-fire.

Forest regions or districts (typically > 100,000 ha) can be stratified according to forest type, soil type, terrain and management practice as a basis for assessing the potential risk to soil values. Such stratification and risk assessment can underpin a strategy to monitor soil change on selected management units. Stratification within the managed (e.g. harvested) area can improve the efficiency of soil sampling and assist interpretation of the importance of measured soil change. A system based on line transects and quadrat sampling is demonstrated for quantifying site disturbance and change in soil bulk density after clearfelling and regenerating tall wet *Eucalyptus regnans* forest. A temporal framework for monitoring soil change that takes account of the contrasting rates of change of different soil properties is proposed. A sample of new areas should be selected for monitoring over time (at about 5-yearly intervals) to capture the effects of changing practices, and to enable an assessment of the effectiveness of 'best management practice'. A strategic approach to monitoring that focuses on representative higher-risk situations, and tests the effectiveness of practices used to mitigate soil damage, can be cost-effective.

The careful application of soil indicator concepts can help improve forest management planning to protect soil values, but more basic research and semi-operational testing of interim approaches is needed to support cost-effective operational monitoring systems in most regions. A better justification for selected indicators, smart approaches to monitoring, and a more solid basis for interpreting change in soil indicators is generally required.

It is important to stress that the baseline against which any soil change should be judged will differ between natural forests and plantation forests which are increasingly being established by afforestation of degraded agricultural land.

To date, there has been much discussion of the theory of how C&I can improve forest management; it is now time to test the practical application of these concepts using 'best-bet' interim approaches. In many jurisdictions, initial indicators are likely to be more 'input' based (e.g. planning guidelines) rather than outcome-oriented, but the goal must be to devise measures of management impacts on important soil processes that can provide the basis for adaptive soil management. To assist this, research, monitoring and forest management need to be closely linked, so that new findings in any one of these areas can be quickly used to improve the effectiveness of the others.

1 Introduction

In recent years the broad concepts of sustainable forest management (SFM) have been widely discussed and accepted. Maintenance of soil fertility is a critical component of SFM because the soil plays a key role in regulating most ecological processes (e.g. rates of energy fixation and forest growth, energy transfers, nutrient cycling, hydrology and water quality, and biodiversity). Forest management practices that retain the stability of the soil are essential to the production of high quality water in forests (Fig. 14.1). In forests managed for wood production, maintenance of productive capacity is a critical component of sustainability, and this is directly linked to the fertility of the soil.

Soil fertility can be defined as the 'inherent' (as opposed to that obtained following inputs such as fertilizer) capacity of the land to support plant growth. Differences in fertility result from differences in soil physical, chemical and biological properties controlling the ability of plant root systems to acquire air, water and nutrients. Because soil fertility is controlled by the interactions between several properties, as opposed to any single factor operating independently, the quantification of soil fertility or its change due to management is difficult. Scientists have searched for simple surrogates of the complex processes controlling soil fertility (e.g. total N content as an index of N turnover, soil pH as a measure of acidification) but without great success. It should be emphasized that 'soil' indicators should reflect important soil properties and processes, but that these may not be correlated with ecosystem functional attributes (e.g. species composition, forest growth) in any straightforward way (Fig. 14.2).

Fig. 14.1. A sub-alpine eucalypt forest in south-east Australia that is managed for conservation of biodiversity and the protection of water catchments. The forest stabilizes steep, high rainfall landscapes. Prescribed burning is used to reduce fuel loads and the risk of wildfires that can result in erosion and the degradation of water quality and other forest values. Relevant indicators are those relating to biodiversity, soil erosion and change in water quality.

Because it is impractical to monitor most soil processes at an operational scale, emphasis has been placed on identifying a set of key soil properties (indicators) that might reflect soil fertility. The challenge of identifying a small number of generic (or robust) and relatively simple indicators of soil fertility is considerable, given the enormous complexity of the soil ecosystem, and the spatial and temporal variation in important soil processes. Despite this, it is broadly accepted that measures of SOM, soil acidity and base status, soil density and erodibility are relevant. Powers *et al.* (1990b) mounted a compelling argument that a measure of SOM and soil porosity may be a useful surrogate for processes controlling water, nutrient and energy balance and flow in soils. They stated:

> Soil porosity principally controls the entry, exit, and internal circulation of liquids and gases, as well as the soil's physical resistance to root penetration. Site organic matter has both chemical and physical influences on soil processes. Chemically, it provides an energy source for the soil biota, as well as a major reservoir for many soil nutrients. Physically, it reduces the erosive force of water and acts as a barrier to soil water evaporation and heat flux. Together, soil porosity and organic matter ultimately affect soil structure through their control of the physical activity of roots and soil animals; wetting, drying, heating and cooling of soil; and the humic stabilization of soil aggregates.

Despite the logic of the approach, the utility of these soil measures has not often been demonstrated. Further, a range of other indicators is likely to be required for specific conditions.

Various international initiatives (e.g. Montreal and Helsinki Processes) have developed forest sustainability criteria and broad sets of indicators, including soil indicators. However, there has not been adequate treatment of the important issues of scale of measurement, logistics of monitoring and interpretation of measured change (i.e. definition of critical or threshold values and 'standards'). In this chapter we address these issues, and provide some guidance for the application of soil indicators in forests. We emphasize that

Forest management activities

⇩

Changed physical, chemical or biological properties of soils

⇩

Changed soil functional processes
(nutrient and water supply, aeration, root growth, biological activities, carbon cycling and storage)

⇩

Ecosystem functional responses
(regeneration, species composition and N fixation, habitat and fauna, water quality, productivity, C storage, wood production)

⇩

Socio-economic consequences
(e.g. forest industry, biodiversity, recreation, water)

Fig. 14.2. Linkages between management, soil properties, soil processes, ecosystem responses and socio-economic consequences in forests.

research and practice in this area is in its infancy, and only guiding concepts as opposed to recipes for application of indicators can be given at this time.

2 Forest management and soil fertility

There is evidence in the literature that adverse soil change induced by forest management can be detrimental to long-term productivity (e.g. Lacey, 1993; Dyck *et al.*, 1994; Folster and Khanna, 1997; Powers *et al.*, 1990a). The following processes can contribute to a lowering of soil fertility:

- Nutrient loss – due to biomass harvesting, burning, soil erosion, leaching and gaseous losses;
- Organic matter loss – due to physical displacement (windrowing, raking), fire or accelerated soil respiration;
- Surface soil loss – due to erosion or windrowing;
- Soil physical damage – profile mixing, compaction and puddling;
- Lowered rates of N fixation by leguminous understorey plants, caused by altered species composition or abundance following disturbance; and
- Change in hydrology and levels of the water table.

Whilst the major impacts are generally associated with forest harvesting and regeneration practices, important changes can also be associated with roading and the extended growing phase of the forest cycle, whether harvesting is conducted or not (Table 14.1).

Key requirements of a soil indicator are the ability to detect important soil change induced by forest management, and capacity to be applied cost-effectively at operational scales (Raison *et al.*, 1997). Monitoring of soil change is relevant at the management unit (e.g. logging area) level, and potential strategies for achieving this are described later.

In order to be helpful in improving forest soil management, indicators must be considered in conjunction with essential monitoring and evaluation

Table 14.1. Potential threats to soil values associated with forest operations.

Forest activity	Threat
Roading	Loss of area, erosion
Regeneration/establishment	Loss of SOM, nutrient loss in fire or by leaching, erosion (loss of aggregate stability and cover)
Growing	Loss of SOM and nutrients from prescribed/wildfire, grazing impacts, erosion, acidification due to acid rain or fertilization, soil pollution
Harvesting	Compaction, redistribution of topsoil, nutrient export, erosion, loss of SOM

SOM, soil organic matter.

procedures. These aspects can be embedded within a forest management system, and provide a basis for continually improving (adaptive) management as shown in Fig. 14.3.

SFM comprises social, economic and environmental components. As described earlier, forest management may affect soil properties, and these changes may subsequently affect soil functional processes. Changed soil properties and processes can modify the functioning of the overall forest ecosystem, with consequences for socio-economic values. Stakeholders need to develop shared management objectives and agreed targets for demonstrating sustainability. The linkage between these factors is shown in Fig. 14.2.

Soil indicators that reflect change in important soil functional processes, with likely flow-on impacts on other ecosystem values would be valuable. These are likely to relate to:

- Soil disturbance (degree and extent of disturbance, exposure and inversion of mineral soil) – relates to soil displacement and erosion risk, tree growth and water quality.
- Bulk density (porosity, soil strength) – relates to soil structure, aeration, hydraulic conductivity and root penetration.
- SOM (also litter, logging residues) – relates to nutrient availability, soil moisture, soil biota and erosion protection.
- Nutrient-supplying capacity – relates to nutrient uptake, and above ground and below ground plant production.
- Soil acidity and base status – relates to carbon storage, nutrient availability, decomposition rates, metal toxicities and root growth.

The stability and productivity of forest stands depend not only on the physical and nutrient-supplying characteristics of the soil, but also on the presence of potentially toxic substances in the plant root zone. Most forest soils are acid to highly acid (pH (salt) < 4–5.5) with very low exchangeable base content, and

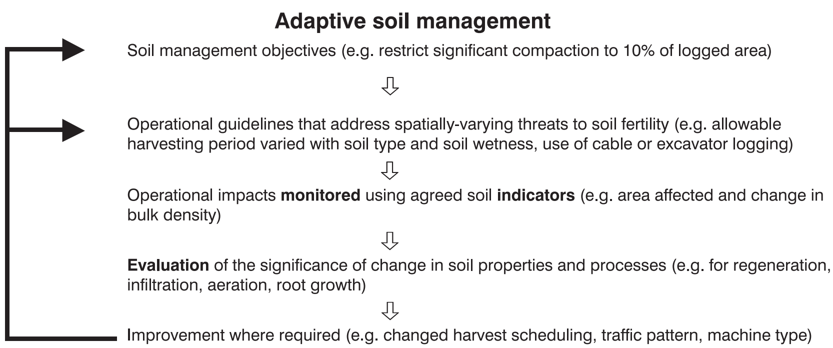

Fig. 14.3. Simplified form of a forest management system, showing how indicators, monitoring and evaluation can contribute to continually improving soil management. The examples used refer to soil compaction.

any management practices that lead to depletion of base cations from soils lead to acidification. Acidification of soils results from the excess net production of protons caused mostly by the discoupling of ion cycles, or from atmospheric deposition of acids and metals.

3 Soil indicators

The following section proposes some field measures for assessing soil change (Table 14.2) and provides a brief commentary for each of these.

3.1 Soil physical change

Soil compaction due to timber harvesting has the potential to affect gas exchange capacity, biological activity, water supplying and holding capacity, and root and tree growth (Burger and Kelting, 1999; Kelting *et al.*, 1999). The measures for quantifying the effect of compaction on these processes, and possible tolerable change (operating standards) are discussed below. It must be emphasized that the soil physical properties limiting to ecosystem function will vary with each ecosystem, e.g. for tree growth, soil strength is important under dry conditions, whereas gas diffusion is important where drainage is poor or bulk density is already high.

Root and seedling growth

Soil bulk density is a measure commonly used to define the potential effect of compaction on root and seedling growth. The threshold values of bulk

Table 14.2. Proposed field measures for important soil properties and functions.

Broad soil indicator	Key functions affected	Field measure
Physical change (4.1e)[a]	Root growth	Bulk density/soil strength
	Aeration	Macro-porosity
	Water movement	Hydraulic conductivity
Chemical change (4.1d)	Nutrient supply	SOM, N&P availability
	Acidification	pH, base exchange
Erosion risk (4.1a)	Many	Area of disturbed soil, ground cover, infiltration rate
Pollution (4.1h)	Soil biology	Accumulation of herbicides, insecticides or air pollutants

[a]Montreal Process (1995) Indicator shown in brackets.
SOM, soil organic matter.

density above which seedling growth will be adversely affected (Greacen and Sands, 1980; Standish *et al*., 1988) could be used as an operating standard. However, the nature of the relationship between bulk density and seedling growth depends on soil texture and plant species (Jones, 1983; Daddow and Warrington, 1984). Early generalizations (Lull, 1959) were that root growth is markedly restricted in fine-textured soil when bulk density exceeds 1.4 Mg m^{-3} and in coarse-textured soils at 1.6 Mg m^{-3}. Further studies with conifer species (Hatchell, 1970; Duffy and McClurkins, 1974; Minko, 1975; Heilman, 1981; Mitchell *et al*., 1982) broadly confirm this range, although some studies (e.g. Foil and Ralston, 1967; Froehlich, 1979) suggest reduced seedling growth at much lower soil bulk densities.

Eucalypts, which have very fine roots, appear to be much more sensitive to high soil density. Williamson (1990), in a glasshouse experiment, showed that *Eucalyptus* seedling weight decreased by 18% when bulk density increased from 0.7 to 0.9 Mg m^{-3}. For a clay loam soil of the Victorian Central Highlands forest, Rab (1994) developed a relationship between bulk density and height or diameter of 3-year-old field-grown *E. regnans* saplings. About 50% reductions (compared to growth in uncompacted soil that had a density of ~0.6 Mg m^{-3}) in height and diameter growth occurred at bulk densities of 0.91 and 0.96 Mg m^{-3}, respectively. For silty loam soil of the same forest, Rab (1996) reported a significant reduction in height and diameter of 3-year-old *E. regnans* saplings growing on primary snig tracks when bulk density exceeded 0.81 Mg m^{-3}.

Froehlich and McNabb (1984) reported that the relationship between change in bulk density and change in height growth of conifer seedlings (*Pinus taeda*, *Pseudotsuga menziesii* and *Pinus ponderosa*) was strong, but the relationship between change in height growth and absolute bulk density was weak. This was probably because the critical bulk density varies with soil texture and tree species as discussed earlier. They proposed that an increase in bulk density of < 20% of the pre-harvest value in the 0–100-mm soil could be used as an interim measure of tolerable change (operating standard).

Roots must overcome the strength of the soil (SS) to penetrate pores smaller in diameter than they are. Since compaction increases SS and decreases macropores, the rate of root elongation and therefore root length can be reduced. The values of SS at which root elongation ceases (critical strength) have been discussed by Greacen *et al*. (1969). These authors summarized soil penetrometer resistance values (SS measured by resistance to penetrometer) ranging over 800–5000 kPa depending on soil type and penetrometer characteristics. The relationships between root growth and SS are poorly known for forest tree species. Zyuz (1968) reported that penetration of pine roots was restricted above 2500 kPa. Sands *et al*. (1979) reported that root development of radiata pine into sandy soils in South Australia is restricted at a penetrometer resistance of 2000 kPa. The relationship between soil strength and growth of *Eucalyptus* roots is not well understood.

Soil strength generally (but not always) increases with declining soil water content, especially in finer-textured soils (e.g. Taylor and Bruce, 1968; Williams and Shaykewich, 1970; Sands *et al.*, 1979; Gerard *et al.*, 1982). Whilst measures of SS made using a penetrometer might be a relatively simple and inexpensive way of monitoring soil physical change, its utility needs to be demonstrated in specific forest types.

Gas exchange capacity and biological activity

Several authors have proposed the use of total porosity and/or aeration porosity as measures of compaction (e.g. Koolen and Kuipers, 1983; Barber *et al.*, 1989; Canarache, 1991). Soil compaction may reduce aeration porosity to the extent that growth or even survival of seedlings is determined by oxygen availability when the soil is wet (Greacen and Sands, 1980). For the Pacific North-West Region of the USA, Howes *et al.* (1983) proposed that more than a 50% reduction in aeration porosity, or a threshold value of 15% or less, were critical values after harvesting. Aeration porosity values below 10% are generally considered restrictive to root proliferation (e.g. Grable, 1971; Greenwood, 1975). This could be taken as an interim threshold value (operating standard) for forest soils.

Soil hydraulic conductivity and water supplying capacity

Infiltration rate (Ir) and saturated hydraulic conductivity (Ks) can be used to quantify the effects of forest practices on soil water movement. Timber harvesting usually reduces Ir and Ks (e.g. Gent *et al.*, 1983, 1984; Incerti *et al.*, 1987; Rab, 1994, 1996) by decreasing the pore-size distribution in the soil. On steep sites, a decrease in Ks may increase overland flow after heavy rain, thus increasing the potential for soil erosion. Thus, Ir and Ks are very good measures of the hydrological behaviour of a catchment. However, their measurement is time consuming and not practical for routine monitoring purposes. Bulk density and aeration porosity may be used as surrogates of Ir and Ks in some situations (e.g. Rab, 1994). Such a relationship should be established for important representative soil types, so as to provide a basis for a simpler monitoring programme.

Reduced Ir and Ks could also restrict water flow from soil to plant root and thus reduce plant growth rates. Little work has been done to determine a critical value of Ks for acceptable plant growth (Gent *et al.*, 1984). However, Boyer and Dell (1980) proposed that a reduction in water infiltration rate in excess of 30% will have a substantial affect on plant-available water and productivity of forests in the Pacific North-West Region of the USA.

In the Victorian Central Highlands eucalypt forest, Rab (1994) compared rainfall intensities with values of Ks and concluded that surface runoff will occur for more than 30% of rainfall events if the value of Ks is reduced to < 5 mm h^{-1}. This corresponds to a reduction in Ks of about 70% and 90% for the clay loam and silty loam soil, respectively, compared to undisturbed soil. As an interim approach, a > 70% reduction in Ks compared to the pre-harvest value in the 0–100-mm soil may be used as an operating standard to quantify the effect of compaction on water movement in this forest. The standard will clearly vary with local conditions.

It is necessary to establish a clear linkage between the above measures of soil physical change and changes in forest ecological processes including growth. As a first approximation for indicator 4.1e (Table 14.2), an interim operating standard might be the proportion (area) of harvested forest with > 20% increase in bulk density compared to pre-harvest values and/or the area with aeration porosity < 10% in the 0–100-mm soil depth (Rab, 1999).

3.2 Soil chemical change

Soil organic matter

As discussed earlier, site (litter plus 0–30 cm depth increments in the soil) organic matter (total organic carbon (C)) may be a useful index of soil fertility and effort should be made to minimize losses relative to the value existing prior to forestry operations. The integrity of the litter and understorey should be sufficiently retained where this contributes to reduced soil erosion (see below).

Organic matter exists in forms ranging from fresh residues to stabilized humic compounds in the soil. There is a range of methods available for separating it into the pools of greatest biological significance (e.g. Stevenson and Elliott, 1989). However, initial emphasis should be kept simple and focus on quantifying total pools, unless research has identified more useful measures for specific ecosystems. The greater cost of more refined analyses will restrict their relevance as indicators for monitoring.

Soil nutrient-supplying capacity

Many forests are limited by N and P availability, so a measure of soil N- and P-supplying capacity would be desirable: it might be possible to achieve this by using regenerating vegetation as a bioassay of rate of nutrient supply, but this requires further evaluation because of the potential for many confounding factors. Maintenance of cation balance may be a problem on specific soil types and should be monitored in those cases. In natural forests, soil N-, P- and cation-supplying capacity should be maintained at pre-harvest values and would be measured 2–5 years after harvesting when the initial effects of major disturbance have passed. Litter and soil C content should be measured at the same time. In plantations, fertilizer is often added to boost nutrient availability at several stages of the crop cycle.

The best way of assessing changes in soil N-supplying capacity is to measure *in situ* rates of N mineralization (e.g. Raison *et al.*, 1992). Sequential soil coring allows estimates of N mineralization, uptake and leaching in the field (Raison *et al.*, 1987). However, these methods are resource demanding and are unsuitable for routine monitoring purposes. At best they could be used on a network of research or selected monitoring sites. There are a number of other methods (e.g. laboratory incubations, chemical extractions, microbial biomass measures) that could be possible indicators of soil N availability (Khanna, 1994), but these are not of generic value and would require evaluation for specific environments. Within a soil type or grouping of related soils there is a reasonable correlation between total soil N or C content and N mineralization rate (Connell *et al.*, 1994). Change in total C or N content will be a much less sensitive indicator of change than change in N mineralization rates, but if calibrated may be useful for more broadscale monitoring.

On a global scale, the availability of P often limits the productivity and health of forests, especially in weathered soils in the tropics and sub-tropics. Like N, only a very small amount of total soil P is available to plants, and changes in total pools are unlikely to be a useful index of the effects of forest management on soil P availability. Changes in the non-occluded inorganic forms of P, and in rates of P mineralization from organic forms, are relevant to rates of P supply to plants. Plants play an active role in the mineralization and solubilization of soil P, and this limits the general utility of many methods of soil analysis for assessing soil P-supplying capacity (Khanna, 1994). Properties of the fine roots of trees such as the initial rate of ^{32}P uptake, and the concentration of P, may be useful indicators of soil P-supplying capacity. Of the soil-based approaches, extraction of labile P by inserting Fe-impregnated filter papers (Menon *et al.*, 1988) in the soil is a potentially useful technique that is simple to use under field conditions.

Soil acidity and base status

A number of soil parameters can be used to define base status and acidity of soils. Base content and soil acidity are inversely related.

1. Soil pH and buffering ranges. Prenzel (1985) described soil buffering ranges associated with carbonate and silicate, aluminium and iron. The transition between these is smooth because changes in cation exchange sites act as an additional buffer. Soil pH may temporarily diverge from the buffer ranges after sudden addition or consumption of protons. Therefore measuring changes in soil pH alone might not be a good measure of soil acidification.

2. Base saturation of soils. Effective cation exchange capacity and the base saturation of exchange sites have been used to define the risk of toxicity due to acidity, and the elasticity of a soil system in relation to change in proton input. Meiwes *et al.* (1986) grouped soils into four categories (Table 14.3) according to their likelihood of toxicity to trees and the elasticity of the soil system to react to changes in proton inputs (further acidification). Corresponding exchangeable cations and soil pH values are also included. The usefulness of base saturation parameters is restricted to reflecting the composition of soil solutions. Base saturation, like pH, is an intensity parameter indicating the direction and the intensity of change but its usefulness in predicting changes is enhanced only by knowing precisely the amount of bases in a soil to a given depth. Changes in the amount of bases in a soil can be assessed by a flux balance approach, but this is a major task and again not amenable to routine monitoring.

3. Ca/A1 ratio in the soil solution. This depends upon the composition of exchangeable cations and is directly linked to the toxicity of Al to plant roots. For example, soil solution with a Ca/A1 molar ratio > 1 is considered to pose a low risk of Al toxicity to spruce roots, whereas a value < 0.1 poses a very high risk leading to extensive injury to fine roots.

4. Chemical composition of the humus layer and fine roots. The humus type and its composition reflect the base status of the forest ecosystem. Meiwes *et al.* (1986) suggested the use of the total Ca to total cation ratio of the O_H-horizon as an indicator of acidification. A base saturation (Ca/Ca+A1+Fe) value > 0.1 suggests very little likelihood of acid toxicity for fine roots and mycorrhizal fungi, values of 0.05–0.1 a medium risk and values < 0.05 a high risk of acid toxicity. The composition of fine roots will reflect the base composition of the soil and soil solution. Meiwes *et al.* (1986) suggested that molar ratios of Ca/A1 < 0.3 in the fine roots in the 0–5-cm depth (A-horizons) and < 0.1 in the 5–20-cm depth would indicate a high probability of acid damage to fine roots.

One can consider three scenarios with respect to soil acidification: (i) where proton input is low; (ii) where fertilizer input is high or nitrate production is high and nitrate is leached after forest harvesting; and (iii) where continuous high inputs of protons occur (acid rain environments). Under low-input

Table 14.3. Classification of forest soils for the likelihood of acid toxicity and their elasticity to further acidification. Cations are expressed as mole(+) fractions of the effective exchange capacity of soils (modified from Meiwes *et al.*, 1986).

Likelihood of acid toxicity	Elasticity	pH-water	Ca + Mg	Al	H + Fe	K and Mg
Very low	Very high	> 5.0	> 0.5	< 0.3	< 0.02	> 0.04
Low	High	4.2–5.0	0.15–0.5	0.3–0.6	< 0.02	0.02–0.04
High	Low	3.8–4.2	0.05–0.15	0.6–0.8	< 0.02	0.01–0.02
Very high	Very low	< 3.8	< 0.005	> 0.8	0.02–0.05	< 0.01

conditions the rate of change in soil pH may be so low that it will be difficult to measure, and is thus of little practical consequence. The best approach for such a system would be to use cation balance (inputs – outputs) and assess the difference in terms of the total base cations on the exchange complex. For case (ii) there may be a short-term change in soil pH (e.g. Khanna *et al.*, 1992). In such cases, changes in base saturation will be a useful indicator of acidification. For case (iii) the choice of parameter will depend upon the nature of the soil and the amount of proton input. Meiwes *et al.* (1986) evaluated a number of soil, humus and plant root parameters which could be used to determine the stability and elasticity of forest ecosystems in relation to soil acidification. They included: soil pH (for grouping soils into proton-buffering categories), base neutralizing capacity, base saturation, Ca/A1 ratio in the soil solution and the ratio of base cations to acid cations in fine roots and humus.

3.3 Erosion risk

Natural forests often have a highly permeable forest floor and surface soil. Forest practices can remove or severely disturb the litter and organic layer and expose the underlying mineral soil (e.g. Bockheim *et al.*, 1975; Rab *et al.*, 1994). This increases the risk of soil erosion by water. Soil erosion is an extremely important issue because soil is a very slowly renewable resource and its loss has a major bearing on water quality, nutrient loss or redistribution, and rate of revegetation. Furthermore, changes in soils, and the consequent effects on vegetation and stream habitats, may ultimately influence the diversity and abundance of fauna in forests. The most critical factors influencing the extent and degree of erosion are site characteristics (e.g. slope, soil type, and climate), inherent soil properties (e.g. water-holding capacity, organic matter content, soil texture, percentage of water-stable aggregates), site management (harvesting prescriptions, especially road and snig track layout, and mitigation treatments) and amount of log removal (e.g. Standish *et al.*, 1988; Lewis and Timber Harvesting Sub Committee, 1991; Rab, 1992). These factors need to be taken into account in developing an overall system for applying this indicator at the forest management level.

A number of surface erosion hazard ratings are currently being used for forested lands, and these have recently been reviewed by Ryan *et al.* (1998). A common one is to use the exposure of mineral soil as an index of soil erosion potential (e.g. Bockheim *et al.*, 1975). In the Victorian Central Highlands, Rab (1994, 1996) found that saturated hydraulic conductivity was significantly reduced (compared to undisturbed areas) on snig tracks, landings and where subsoil was disturbed by harvesting. Based on saturated hydraulic conductivity and exposure of mineral soil, it can be inferred that roads, snig tracks, landings and disturbed subsoil areas are the major sites of soil erosion. Thus, a measure of soil erosion risk can be gained by summing the area occupied by

Table 14.4. Suggested basis for stratifying harvested coupes[a] that can then be used as a basis for sampling soils to determine any change in properties; not all categories exist in the field, and not all are equally important for monitoring soil change.

A: Operational categories		
Unharvested area (UA)	Areas of retained forest or other vegetation	
Harvest area (HA)	General logging area within which trees are felled	
Firebreak (FB)	Perimeter boundary of harvested area	
Snig tracks (ST)	Tracks created by towing or winching logs to the landing	
Log landings (LL)	Area to which logs are snigged for sorting and loading for transportation	
Access roads (AR)	Temporary roads used during the harvesting operation	
B: Soil disturbance categories		
Degree of soil profile disturbance	Type of mixing/ removal	Main horizon
Undisturbed (S0)	Forest intact (FI)	O1
	Understorey intact (UI)	O1
	Litter layer intact (LI)	O1
Lightly disturbed (S1)	Litter layer disturbed (LD)	O2
	Litter layer partially removed (LR)	O2
Moderately disturbed (S2)	Litter completely removed and topsoil exposed (LR)	A
	Litter mixed with topsoil (LM)	A
	Topsoil disturbed (TD)[b]	A
	Topsoil partially removed (TP)	A
	Topsoil mixed with subsoil (TM)	A
Severely disturbed (S3)	Topsoil mixed with subsoil (SM)	B
	Subsoil disturbed (SD)[c]	B
	Subsoil partially removed (SP)	B
	Subsoil mixed with parent material (SC)	C
	Subsoil removed and parent material exposed (SR)	C
C: Soil and slash pile categories		
Soil piling (SP)	Soil piled to a height > 0.3 m	
Soil and slash piling (SS)	Soil and slash piled to height > 0.3 m	
Slash and/or bark piling (SB)	Slash and/or bark piling to height > 0.3 m	
D: Fire intensity		
Unburned (F0)	Litter, soil or vegetation unburned	
Low (F1)	Partial burn of slash and litter of diameter up to 20 mm. Litter O2 horizon, where present, predominantly unburned	
Moderate (F2)	Near-complete burn of slash and litter of diameter up to 20 mm, partial burn of branches greater than 20 mm. Some soil oxidation, but generally charcoal or white ash-bed	

Table 14.4. cont'd.

High (F3)	Near-complete burn of slash and litter of diameter up to 70 mm, partial burn of branches greater than 70 mm. Soil oxidation (orange ash-bed) predominant
E: Cover (%)	

[a]For each soil indicator a different set of categories may be sampled (see Table 14.7).
[b]Topsoil consists of A_1, A_2 and A_3 horizons except where A_2 is conspicuously bleached whereby A_2 and A_3 are regarded as subsoil.
[c]Subsoil includes B_1 and B_2 horizons and conspicuously bleached A_2 horizon (and then any other A-horizon below the A_2).

access roads, firebreaks, log landings, snig tracks and subsoil-disturbed harvest areas (HA-S3) (Table 14.4).

It is not possible to specify standards for soil erosion, but ideally erosion rates should not be accelerated from the coupe, and adequate precautions should be taken to minimize soil redistribution down slope within the harvested area. Attention must be paid to sheet erosion, gullying and mass movement. In the absence of a standard, we advocate the consideration of best practice as specified in guidelines based on definition of erosion hazard class. Erosion hazard is a function of soil erodibility (inherent soil properties), erosivity (rainfall characteristics) and slope. Erodibility can be estimated from geology (least desirable), soil type or soil properties (better estimate). The erosion hazard is used to determine various management constraints (e.g. slope limit for logging, frequency of cross drains on snig tracks), and whilst the approach is reasonable, there is a clear need to test the adequacy (i.e. calibration with soil erosion rates) under field conditions using a strategy of monitoring and research.

4 Monitoring

To be able to apply soil indicators operationally, a framework that includes spatial and temporal aspects is required. This can provide the basis for developing an overall system for quantifying soil change at the forest management unit (FMU) level, and is discussed in the following sections.

4.1 Regional or district level framework

The forest region or district (size will vary but may typically be 100,000 ha or more) may be stratified and the areas where there is potentially a high risk to soil values can be identified by considering factors such as those presented in

Fig. 14.4. This framework was applied, as an example, in two forest districts in Victoria to determine the number of coupes (logging areas) to be monitored (Table 14.5). It was found that sampling four or eight logging coupes would sample the higher risk environments in each district. The sampling would

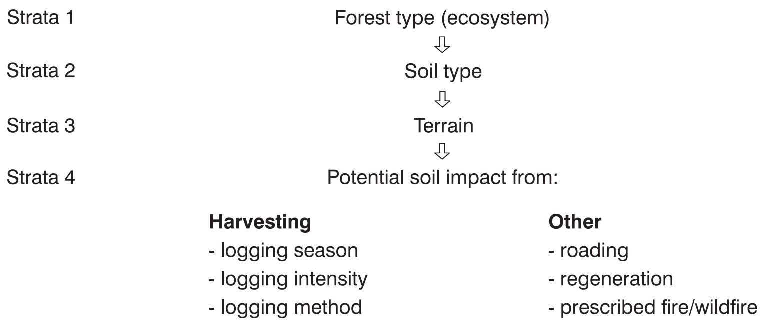

Fig. 14.4. A spatial framework (stratification) for assessing risk to soils and to guide development of a strategy to monitor soil change at the forest management unit level. By working down the strata, the likely risk of soil damage at a particular site under a specific management regime can be determined. Priority can then be given to monitoring potentially high risk situations.

Table 14.5. Summary of forest environment and management systems for two forest districts in Victoria, Australia. Sites with high risk of soil damage are given, and the number of operational coupes suggested for monitoring.

	Noojee forest district (area ~100,000 ha)	Orbost forest district (area ~200,000 ha)
(i) Description		
Dominant forest type	Mountain ash (*Eucalyptus regnans*)	Mixed species
Parent material/soil type	Granitic	Tertiary, sedimentary and granitic
Terrain	Steep, gentle	Steep, gentle
Logging intensity	Clear-felling system	Seed-tree system
Logging season	Mainly summer logging	Mainly summer logging
Machinery configuration	Skidder/dozer	Skidder/dozer
Regeneration techniques	Mainly slash burning	Mainly slash burning
(ii) High risk site categories	Steep sites, dozer or skidder logging	Steep sites, tertiary and granitic soils, dozer or skidder logging
(iii) Coupes to be monitored[a]	4	8

[a]Based on two replicate coupes for each high-risk category. A new set of coupes may be selected for monitoring at approximately 5-year intervals to capture the effects of change in operational practice.

need to be repeated over time. The type of indicators selected and the basis for interpreting the importance of change in them will vary between the two Victorian examples shown. The mountain ash forest grows on deep, permeable and fertile soils, whereas soils in the Orbost region are of much lower chemical fertility, and some (e.g. granitic types) are erodible.

4.2 Site level framework

Stratification of harvested coupes

The type of stratification developed will depend on the number of clearly defined disturbance categories that exist on the forest site following management operations. In some cases there may be no advantage in stratification prior to sampling, but in heavily disturbed sites with contrasting soil conditions there clearly is. Major soil disturbance occurs during clear-fell harvesting and slash burning in tall *E. regnans* forest in southern Australia (Table 14.6). A stratification scheme for dealing with this has been developed (Table 14.4) and as an example its application is described.

This stratification system was applied to 25 operational logging coupes (each from 10 to 40 ha in area) in *E. regnans* forest of the Central Highlands of Victoria. The coupes were sampled using a transect method. On each transect (50 m spacing) a 1-m^2 quadrat was studied at 10-m intervals to identify operational areas, dominant horizon and level of soil mixing and removal (Table 14.4). These results showed that S0 ranged from 15 to 45% of the coupe area, averaging about 29% (Table 14.6). S1 ranged from 0 to 15%, S2 ranged from 24 to 61%, and S3 ranged from 7 to 32% of the coupe area. In all logging

Table 14.6. Fraction of coupe area affected by combinations of operational category and degree of soil disturbance (see Table 14.4 for descriptions) after clear-felling of Victorian mountain ash (*Eucalyptus regnans*) forest.

	Soil disturbance				
Operational category	S0	S1	S2	S3	Total by operation
UA	0.5 ± 0.2	0	0	0	0.5 ± 0.2
HA	29.1 ± 1.8	3.3 ± 0.6	33.9 ± 1.8	7.7 ± 1.1	74.0 ± 1.1
ST	0	0	11.0 ± 1.3	7.3 ± 1.0	18.3 ± 1.1
FB	0	0	1.4 ± 0.4	0.7 ± 0.3	2.1 ± 0.5
AR	0	0	0.4 ± 0.2	1.3 ± 0.4	1.7 ± 0.4
LL	0	0	1.3 ± 0.3	2.1 ± 0.4	3.4 ± 0.5
Total by disturbance	29.6 ± 1.7	3.3 ± 0.6	48.0 ± 2.3	19.1 ± 2.0	

Values (%) are the mean of 20 logging coupes ± SE of the mean.

coupes, the most common category was S2 (accounting for 44–81% of the total disturbed area), followed by S3 (10–58%). The majority of the disturbance occurred within the general harvest area (HA, Table 14.5). The HA varied from 64 to 83% of the coupe area, snig tracks occupied 12–31%, firebreaks 0–7%, access roads 0–5%, and log landings 1–7% of the coupe area.

Clearly, not all disturbance categories are equally important either in terms of spatial coverage or soil change. To determine which category is relatively most important, in terms of impacts, several workers have related various classes of soil profile disturbance to changes in soil properties, soil erosion and tree growth (Miller and Sirois, 1986; Farrish, 1990; King *et al*., 1993a,b). In the *E. regnans* forest, Rab (1994) found significant increases in bulk density of topsoil and subsoil on disturbed harvest areas (HA-S2, HA-S3, Table 14.4). King *et al.* (1993a,b), in the same forest, showed that height and diameter growth of regenerating *E. regnans* 3 years after sowing was not significantly different on topsoil-disturbed harvest areas compared to those grown on undisturbed areas, but they found significantly reduced growth on subsoil-disturbed harvest areas (HA-S3). Further research is needed to determine the acceptable level of soil mixing or removal in this forest. However, as an interim approach, areas affected by AR, LL, FB, ST and HA-S3 (see Table 14.4 for definitions) may be taken as areas where soil properties are likely to be adversely affected by harvesting and for which temporal change should be quantified. These represent about 33% of the coupe area in the mountain ash forest (Table 14.6). Snig tracks and HA-S3 comprise most of this and are thus a priority for measurement (Rab, 1999).

Table 14.7. Suggested coupe-level sampling strategy for monitoring a range of soil properties following harvesting. Nomenclature follows that given in Table 14.4; see Table 14.9 for suggested sampling frequency.

Indicator	Method	Priority sample points[a]
Bulk density	Transect-quadrat	LL, ST, HA-S2 and S3
SOM	Transect-quadrat	All
Acidity	Transect-quadrat	May require lower sampling intensity
Pollution	Transect-quadrat	May require lower sampling intensity
Erosion		
Soil disturbance index	Transect-quadrat	All
Cover	Transect-quadrat[b]	All
Aeration porosity	Stratification	LL, ST, HA-S3
Hydraulic conductivity	Stratification	LL, ST, HA-S3
Nutrient supply	Stratification	Areas of likely major change

[a]Initial studies may be needed to establish the most important areas in specific forest systems. This table provides a guide only.
[b]Base the year 2 assessment on a stratified approach.
SOM, soil organic matter.

Monitoring for a range of soil properties

As discussed above, not all soil disturbance categories exist in the field, and not all are equally important for monitoring soil change. The following strategy is proposed to monitor soil properties in forests where soils are highly disturbed during harvesting or regeneration operations (Table 14.7).

As an example, changes in bulk density in the surface soil (0–100 mm) immediately after timber harvesting and slash burning on one coupe in the mountain ash forest are shown in Table 14.8. Transects were located 50 m apart and samples taken at intervals of 10 m, giving a total of 195 pre-harvest and 196 post-harvest samples. These findings again demonstrate the importance of sampling the S2 and S3 parts of the HA and other heavily disturbed areas. The need to sample less area of the coupe makes monitoring a more tractable exercise.

4.3 Temporal framework

The rate of change or recovery of soil properties is variable, and this must be taken into account when designing soil monitoring regimes. In general the recovery of the area affected by soil profile disturbance and regeneration fire is usually faster than the recovery of compacted and eroded soil. For example, the recovery of compacted forest soils, in the absence of ameliorative treatment, is

Table 14.8. Effects of harvesting and slash burning on change in soil bulk density (Mg m^{-3}) in the 0–100 mm soil depth on one logging coupe in mountain ash forest in Victoria (mean and standard error are shown, with number of samples in brackets).

		Post-harvest			
		Soil disturbance category			
Operational category	Pre-harvest S0	S0	S1	S2	S3
UA	0.558 ± 0.009 (195)	0.580 ± 0.036 (6)			
HA		0.528 ± 0.038 (12)	0.566 ± 0.039 (7)	0.623 ± 0.012 (128)	0.718 ± 0.053 (19)
ST				0.783 ± 0.027 (10)	0.973 ± 0.042 (5)
FB				0.884 ± 0.083 (3)	1.069 (1)
AR					1.198 (1)
LL				1.099 ± 0.093 (2)	1.348 ± 0.079 (2)

usually slow under the influence of climatic processes and the activity of roots and soil fauna. Shoulders and Terry (1978) reported that soil porosity changes resulting from site preparation persisted up to 6 years. However, it may take 10–20 or more years for soil to recover after shallow compaction (Dickerson, 1976; Froehlich, 1979; Jakobsen, 1983), while compaction of deeper layers may persist for 50–100 years (Greacen and Sands, 1980). Rab *et al.* (1992) showed that bulk densities were still significantly greater on snig tracks and landings compared to undisturbed areas 25 years after harvesting in the East Gippsland forest in southeastern Australia. In *E. regnans* forest in the Central Highlands, Jakobsen (1983) found that bulk densities on the primary snig tracks were significantly greater than on nearby undisturbed soil 32 years after harvesting. It is not possible at this stage to determine how long it will take for the affected soils to recover.

A temporal framework for monitoring a range of soil properties is proposed (Table 14.9). Many soil variables should be measured soon after major forest disturbance, but not nutrient-supplying capacity which usually shows ephemeral increases. Ground cover and nutrient availability can show rapid temporal change and should be measured more often for 2–5 years after disturbance. Bulk density, SOM and nutrient-supplying capacity could then be remeasured 30 years after disturbance and at the end of the rotation. Soil acidity and pollution might be measured only infrequently.

Table 14.9. Suggested temporal framework for monitoring indicators of change in soil properties/functions.

	Monitoring interval (years after disturbance)[a]				
Indicator	0	2	5	30[b]	Other
Bulk density	✓			✓	End rotation
Aeration porosity	✓				
Hydraulic conductivity	✓				
Erosion					
Soil disturbance	✓				
Cover	✓	✓			
SOM	✓			✓	
Nutrient supply[c]	?	✓	✓		?
Acidity					Every 30 years?
Pollution					Every 30 years?

SOM, soil organic matter.
[a]Repeated each 5 years for a new sample of disturbed areas (e.g. harvested coupes).
[b]In short-rotation plantations, measures would be taken at the end of the rotation, rather than after 30 years.
[c]Measure of N and P availability.

A sample of new areas should be selected over time (at, say, 5-yearly intervals) to capture the effects of changing forest practices and to allow an assessment of the effectiveness of so-called 'best management practice'. Temporal sampling of these areas is needed to quantify change in soil function (Table 14.9). The aim is not to generate a comprehensive coverage of soil change for the whole forest area (this would be too expensive), but to use strategic sampling to focus on high-risk situations, and to test the effectiveness of practices used to mitigate soil damage. A representative sample of disturbed areas, based on the spatial sampling strategies proposed above (Tables 14.5 and 14.7), should be monitored. Thus only a sample of areas, selected from a portion of a forest district, would be monitored; and the frequency of sampling on these areas minimized (Table 14.9). Such an approach makes monitoring soil change feasible.

5 Evaluation

Improved (adaptive) soil management requires an evaluation of the significance of any measured change (trend) in soil properties or processes (Fig. 14.2). Ideally the evaluation should be against (or guided by) a performance measure or target that is defined in a forest management plan. The scientific basis for the evaluation process is probably the most poorly developed aspect of the set of linked activities required for the application of indicators for adaptive forest management. Performance measures will vary according to management objectives, spatial scales and ecological conditions. The specification of performance measures or targets is a critical step in forest management planning. Ideally these should be formulated after discussion between interested parties (stakeholders).

Performance measures or targets can take a variety of forms, some of which are:

- Standards. These are measurable parameters established for use as a rule or basis for comparison in measuring or judging quantity, quality, value, capacity or other characteristics (Maini, 1993). A few scientifically based standards exist in forestry (e.g. those for water quality), but in general these are still poorly developed.
- Interim standards. These are generally based on the balance of scientific opinion and are adopted until better scientifically-based standards become available. A key issue is the calibration of 'standards' for specific forest ecosystems (e.g. establishing threshold values for forest properties and processes). This requires research, and commonly examines the effects of alternative management practices, including amelioration, on important soil properties and processes affecting the forest management objective for a given area of forest.

- Compliance with plans, guidelines and prescriptions. This is the approach most commonly adopted, but it generally focuses more on inputs rather than on outcomes (soil impacts) and thus is not a satisfactory approach in the long term.

The evaluation step needs to consider three important aspects (see, for example, Burger and Kelting, 1999; Kelting *et al*., 1999):

- The spatial portion (area and distribution) of the management area where significant soil change occurs. Whilst the negatively impacted area should clearly be minimized, the tolerable magnitude of area is generally set arbitrarily and refined over time. Current 'best practice' and economic trade-offs are often considered in setting the tolerable level of change.
- The pattern of temporal change for degradation, recovery or improvement (e.g. after afforestation) in soil values. This is particularly important for variables such as erosion risk, or soil nutrient-supplying capacity that can show high temporal variation.
- Integration of the consequences of a variable change in several soil properties. This can be addressed by computing a soil quality index (based on critical levels of soil properties) in relation to important ecosystem attributes such as forest productivity (Kelting *et al*., 1999).

To date very little work has been done on the difficult topic of defining performance measures. A useful initial approach to defining operating standards for soil indicators has recently been proposed by Rab (1999) for *E. regnans* forest in the Victorian Central Highlands. The exercise highlights the major sources of detrimental soil change during harvesting operations and methods for minimizing soil impacts.

6 The future

Monitoring of change in soil properties and processes is rare in operational forestry. Generally guidelines for protecting soil resources are specified in codes of forest practice or in harvesting plans, but the effectiveness of these in protecting soil fertility is generally not assessed in any quantitative way. Judicious application of soil indicators is a step towards achieving improved (adaptive, Fig. 14.2) soil management. As already discussed, a better justification for selecting indicators, smart approaches to monitoring, and a more solid basis for interpreting change in soil indicators is required. Basic research and semi-operational testing are needed. In most jurisdictions, operational application of soil indicators is still some way off. Case studies in important representative forest types would be an effective way of evaluating interim soil indicators and developing approaches for applying them at operational scales. The case studies need to be nationally coordinated in order to gain maximum benefit.

Case studies would be particularly valuable in defining the level of precision of soil measures that is feasible (cost-effectively obtained) for particular forest systems. Variability can be high in natural forests and this can increase further following major disturbance such as harvesting. Trade-offs will be required between detailed measures on fewer sites and more approximate estimates for a greater range of sites.

Research, monitoring and forest management need to be closely linked, so that new findings in any one of these areas can be quickly used to improve the effectiveness of the others. Further, well-designed field experiments are essential for understanding the long-term effects of alternative forest management practices on soil fertility. These provide the basis for improvements to practice (e.g. Risser, 1991; Leigh and Johnston, 1994).

An important issue is how to stratify the forest landscape so that representative sites can be located for monitoring or research, and to provide a mechanism (spatial model) for applying the findings from these activities to improve broad-scale forest management. This issue is especially important in relation to forest soil properties because these have a major influence on forest sustainability and can be markedly modified by forest management, and because soil surveys useful for guiding management are still relatively rare in most parts of the world.

Recently, methods have been developed to predict at landscape scales the spatial distribution of soil properties important for forest management (e.g. Gessler *et al*., 1995; Ryan *et al*., 1996). The landscape can be stratified according to geology, landform and climate, and variables derived from these. Spatial modelling can then be used to help identify landscape types and locations that are most sensitive to change (e.g. compaction, erosion) and where monitoring or research should be focused.

Selection of soil survey, research or monitoring sites may also be assisted by the use of airborne gamma-ray spectrometric data which provide information related to the chemical composition of the upper 30–50 cm of the soil profile (e.g. Cook *et al*., 1996). Such data may allow a more precise stratification than that possible from geology, and has the advantage of providing a complete spatial coverage of any area that has been surveyed. The utility of this methodology requires broader application and testing.

Clearly, new approaches can greatly assist the collection and use of soils information to improve forest management. However, the capacity to apply these remain limited in many parts of the world.

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15

Catchment and Process Studies in Forest Hydrology: Implications for Indicators of Sustainable Forest Management

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Manipulation of forest cover will influence the quantity and timing of delivery of streamflow. Changes in evaporation due to change in forest cover is the cause of modifications to streamflow. Evaporation from forests is comprised of two different components; evaporation of intercepted rainfall and evaporation of water taken up by the roots through the process of transpiration. The magnitude of the impact of changes in forest cover will depend on the balance between the two components. A few years after forest harvesting, vigorous regrowth can mean that streamflow may decline to below levels that existed before forest manipulation. There may also be changes in the timing of streamflow by modification of forest cover. This may be because different soil water extraction regimes influence base levels of stream behaviour but also the dynamics of flow will be impacted by the effects of forestry activities such as road construction and use, and harvesting activities.

Traditional methods of gauging streamflow are costly, long-term enterprises whose precision may be insufficient to detect the impacts of small changes in forest cover. Alternatives may be simplified gauging methods, empirical relationships with forest cover or use of available models of varying complexity and data requirements. The most sophisticated of models will be limited by the availability of information about evaporation characteristics of unfamiliar tree species and soil physical properties, particularly in remote tropical areas. Because high water use and vigorous growth are linked, an alternative which can serve as an indicator is the rate at which the forest stand is growing. This can be gauged from growth increments, traditionally acquired by forest mensuration.

Increases in sediment production during forest harvesting are a major contribution to the deterioration of water quality from disturbed forests. Of paramount importance are the contributions made, particularly, to the suspended sediment load of streams from the construction and use of tracks and roads. The residence time of sediment in catchments is also an important consideration. The residence time can be of the order of years and impacts of forestry activities may not be experienced downstream until a major flow event dislodges stored sediment.

Chemical properties of streams draining from undisturbed forests vary in relation to geology and annually in relation to rainfall. Impacts of forest disturbance are greatest in forests with the most closed nutrient cycles. Physical and chemical properties of stream water can be monitored automatically but in some cases resources may not be adequate to sustain such a sampling programme. Occasional point sampling may be an alternative. However, no means of sampling gives a direct indication of the impacts of forest activities on stream biota resulting from changes in the physical and chemical characteristics of the aquatic habitat. Biological monitoring of indicator groups of species can indicate the quality of the stream habitat. Macroinvertebrates and fish, separately or together, are favoured indicator groups of the quality of the stream habitat. Biological monitoring has been little practised in relation to forest management. There is a need to improve information about candidate indicator taxa for undisturbed forest habitats at relevant (sub-national) scales.

1 Introduction

A major concern for sustainable forest management (SFM) is to maintain the value of forests in relation to services, such as the amount and quality of water that forests provide both within and outside of the forest. The variety and scope of water services or values is very large, ranging, for example, from a pristine aquatic habitat for various plants and animals within the forest to providing adequate stream and river discharges for navigation and reservoirs far distant from the forest boundary.

The manipulation of forest cover, either by harvesting, planting new forests or deforestation may all influence the quantity and quality of water emanating from the landscape. For a large part of the last century, hydrologists have been examining the impact of land use changes on streamflow and water resources in catchment studies. Increasingly, process studies have been undertaken, often nested within catchment studies but operating at the plot scale. Process studies have attempted to evaluate the detailed mechanisms controlling, particularly, the amounts and movement of water, solutes and soil particles in forest lands and how these are affected by manipulation of forest cover. Very often these investigations have been conducted as long-term studies involving substantial human and technical resources. By now, therefore, there is accumulated information about the impacts of forest cover on water yield and quality of water from catchments representing at least some forest areas.

The debate about sustainability indicators provides a new challenge for those with experience in the many aspects of forests and their interactions with the quantity and quality of water. Familiar questions are being asked which hydrologists should be well placed to at least attempt to answer. What are the best means to evaluate the different water services provided by forests? What are typical values for important hydrological parameters? By how much do these values naturally fluctuate? What levels of impact are needed before any change can be detected with some certainty?

This chapter attempts to do two things. Firstly, it attempts to demonstrate how soundly such questions might be answered by reviewing a range of results from studies in which the forest cover has been manipulated. Secondly, it draws on the information available to propose indicators of SFM relevant to fluctuations of water quantity and quality.

This chapter does not provide details of the instrumentation, methods and specific techniques which are referred to. This information can be found in the relevant cited references but also for example in Maidment (1993). Neither is it the aim of this chapter to tender specific technical guidance on various aspects of forest management in relation to various aspects of water quantity and quality. For temperate forests this information might be sourced from Satterlund and Adams (1992) and Brooks *et al.* (1997), while references with a tropical perspective may be found in Bruijnzeel (1990, 1997).

2 Forests and water quantity

2.1 Forest hydrology

Traditionally, the impact of vegetation on water yield from land areas is assessed is by establishing catchment or watershed studies. The essential features of these are that rainfall, representative of the catchment area, is measured as is the streamflow or runoff which is gauged by a weir or flume. In the simplest case it is assumed that on an annual basis there is little change in water storage in the catchment, and, also assuming no leaks from the catchment, the difference between annual rainfall and runoff is the evaporation from the vegetation covering the catchment. The impacts of forest management might be manifested as changes in the amounts and timing of streamflow. Figure 15.1 shows the annual rainfall and streamflow over an extended period for an undisturbed forest at Hubbard Brook, New Hampshire, USA (Likens and Bormann, 1995).

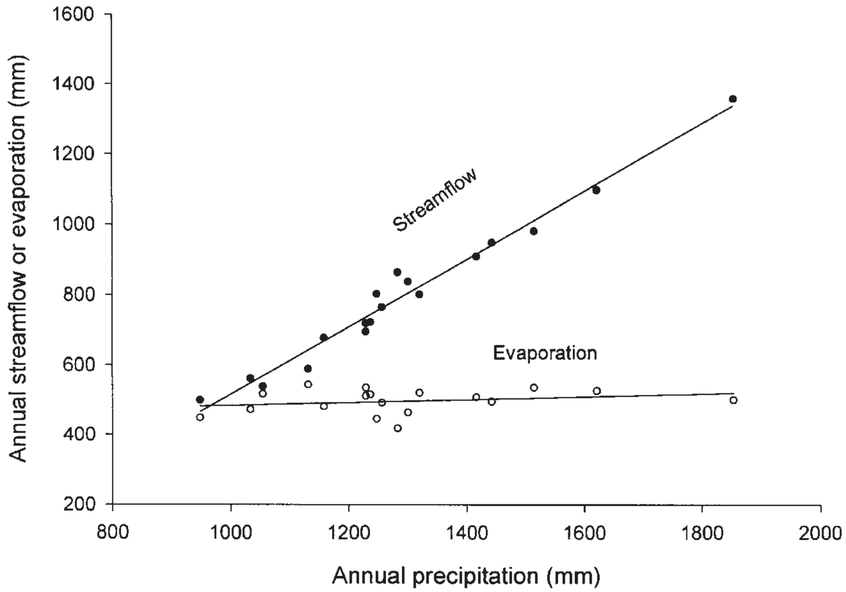

Fig. 15.1. Relationship among precipitation, streamflow and evaporation for mixed hardwoods at the Hubbard Brook Experimental Forest during 1956–1974 (after Likens and Bormann, 1995).

There is considerable year to year variation in rainfall which, in this case, is closely matched by fluctuations in the streamflow. There is a strong positive relationship between rainfall and streamflow and there is an almost constant difference between rainfall and streamflow, the annual catchment evaporation. The evaporation is virtually constant at around 500 mm year^{-1} even though rainfall spans a range from 800 to 1800 mm. This probably means that soil water deficits rarely if ever limit transpiration loss and that factors come into play to maintain transpiration at a constant level. The negative feedback between stomatal conductance and air humidity deficit has a big part to play in limiting transpiration and maintaining it constant from day to day (Roberts, 1983). Rainfall interception for a broadleaf woodland, such as that at Hubbard Brook, might typically be 15% of the gross rainfall. Over the range of annual precipitation observed, interception loss might span 120–270 mm year^{-1}. However, in wet years there may be a reduction in transpiration because of the greater fraction of time the canopy is wet. Overall then an increase in evaporation due to enhanced interception loss may be quite small. In Fig. 15.1 the regression line between rainfall and streamflow might be used as a calibration against which the effects of any forest management, which impacts the rainfall/runoff relationship, are judged. Nevertheless it should be considered how many years might constitute a good calibration period and that the good relationship observed in Fig. 15.1 also reflects a substantial financial investment in various resources to ensure a quality output. Calder (1992) has discussed some of the errors associated with catchment experiments and believes that at least 2–3 years of data are necessary to detect an effect as long as errors are only random and not systematic. Calder (1992) also advises that, prior to establishing an experiment, a careful study be made of the likely systematic and random errors of measurements and the numbers of years of data that will be required to measure the anticipated effect within the accuracy of the experimental data.

The simple situation described above is probably representative of temperate broadleaf forests in which no significant soil water deficits occur, but other examples show at least two key differences from the Hubbard Brook example. One difference is observed in catchment studies in which increases in annual evaporation are associated with increases in annual rainfall, suggesting that soil water availability limits forest water use. This means that it is important that any considerations of the impacts of forest management should be made with knowledge of the whole rainfall range before considering the impacts of the manipulation of the forest cover. The effects of fluctuations in rainfall can have quite different effects on the behaviour of streamflow in different forest regions. Studies in the jarrah (*Eucalyptus marginata*) forests of Western Australia (Ruprecht *et al.*, 1991) showed increases in evaporation with increasing rainfall but there was no concomitant increase in streamflow (Fig. 15.2). The result implies that any additional rainfall is evaporated by the trees, understorey or from the soil, while there is no supplement to streamflow. A linear increase in evaporation with annual rainfall was also observed by Blackie (1979) in tropical rainforest in western Kenya but in this case there was also an increase in streamflow as well as evaporation associated with increasing rainfall (Fig. 15.3). A second problem for catchment studies is the effect of year to year differences in catchment storage on the interpretation of streamflow records. Blackie (1979) found differences in soil and groundwater storage of up to 250 mm in the catchment from year to year. Any assessments of the impacts of forest management will need to take account of the levels of storage before the true effect of manipulation of the forest on streamflow can be evaluated. Results such as those from the Blackie (1979) study in which there are differences in catchment storage from year to year suggest that over a number of years the effect on estimates of streamflow from the catchment will be minimized and places further emphasis on the point made earlier that only after an absolute minimum of 2–3 years will the average catchment behaviour begin to emerge.

2.2 Forest cover and annual streamflow

Analysis of the results of many catchment experiments involving forest clearance or afforestation have been made by Hibbert (1967), Bosch and

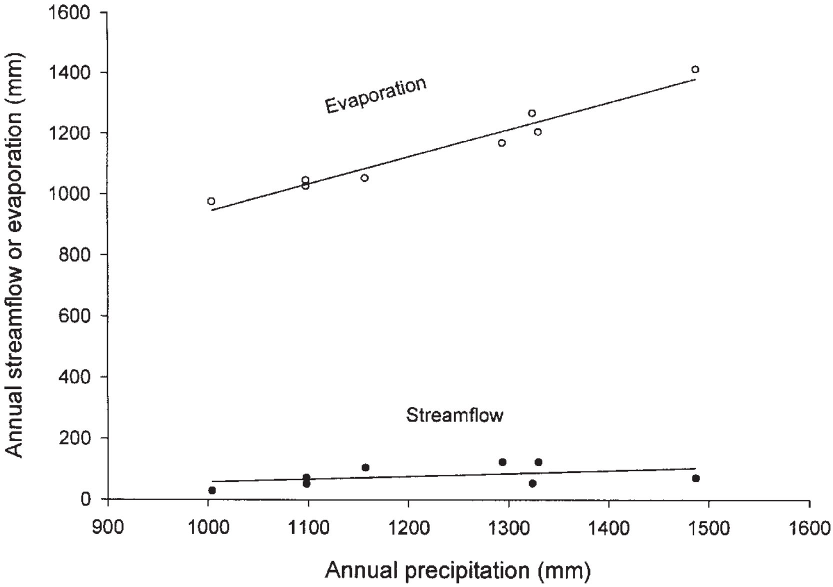

Fig. 15.2. Relationship among precipitation, streamflow and evaporation for undisturbed jarrah (*Eucalyptus marginata*) forest, Western Australia during 1978–1985 (after Ruprecht *et al.*, 1991).

Hewlett (1982) and Stednick (1996). The conclusions are that reduction of forest cover increased water yield and afforestation decreased water yield. However, the responses to treatment were highly variable and unpredictable. Figure 15.4 shows the data assembled by Bosch and Hewlett (1982) to support the conclusion that water yield increases proportionately with the forest cleared. There is a great deal of scatter which the reviewers related to species differences. The trend lines in Fig. 15.4 are fitted to the data for conifers and eucalypts, and for deciduous forests. Bosch and Hewlett (1982) hypothesized that vegetation growth rate was a key control of the likely response of a catchment to disturbance. Figure 15.4 shows that each 10% reduction of forest cover results in an annual yield increases of 40 mm for conifers, 25 mm for deciduous hardwoods and 10 mm for scrub (not shown in Fig. 15.4). Bosch and Hewlett (1982) make the point that removal of less than 20% of tree cover is unlikely to be detected statistically by streamflow gauging.

Stednick (1996) examined information from 95 paired catchment studies in the USA. The yield increases associated with forest clearance are similar to those given by Bosch and Hewlett (1982) but with even more scatter. He showed that streamflow would increase by 25 mm for each 10% of forest area removed. He also concluded that on average 20% of forest would need to be removed before an effect could be detectable but this varied considerably for individual regions in the USA, being as little as 15% in the Rocky Mountains

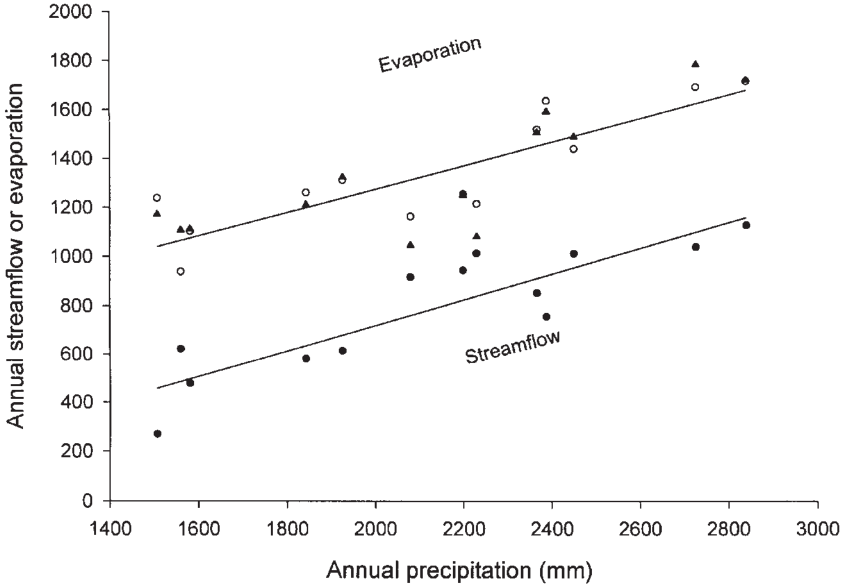

Fig. 15.3. Relationship among precipitation, streamflow and evaporation for undisturbed tropical rainforest at Kericho, Kenya during 1960–1973 (after Blackie, 1979). Solid triangle symbols represent annual differences between rainfall and streamflow with corrections for storage in soil and groundwater.

to 50% in the Central Plains. From a substantial number of studies in the north-east USA, Hornbeck *et al.* (1993) showed that the first-year increase in streamflow per 10% of forest cleared ranged from 12 to 35 mm. These authors emphasized felling patterns and clearance methods as factors influencing catchment response. They observed that streamflow increases were greatest in catchments from which the forest was cleared from the valley bottom runoff generating zones. Presumably in a catchment these are the areas suffering least from soil moisture deficits, and the trees may also benefit from an accumulation of nutrients in these stream side zones.

Much of the data that has been reviewed in the studies above is based on catchment experiments in which forest cover is partially or completely cleared. Regeneration of the forest may begin soon afterwards and this may cause problems of interpretation. Immediately following clearing the greater soil water deficits under forests will be satisfied which may mean that there is an underestimate of forest clearance effects, particularly in the first year or so following clearing of forest. Modification of the soil surface by logging activities will enhance runoff thereby overestimating the effects of forest clearance effects. Rainfall variability in the few years following clearance is also a consideration and could lead to an under- or overestimation of the effect of forest removal.

The section above on the effects of deforestation on streamflow dealt with short-term effects immediately following the treatment and in the 2–3 years following. Generally, regrowth of vegetation will take place in some form which will probably render the initial streamflow changes as only temporary. Hornbeck *et al.* (1993) reviewed results from a number of long-term catchment studies in the north-east of the USA and concluded that streamflow increases rarely last for more than a decade. The increased streamflow following deforestation could be maintained, however, by further cuttings or herbicide treatments. Cornish and Vertessy (2001) showed that in eastern Australia streamflow increases peaked 2–3 years after clearance and returned to pre-treatment levels in a period of between 4 and 8 years. They observed that the rate of decline in streamflow from the peak increase was proportional to the stocking rate of the regenerating forest. It would be expected that streamflow increases, following forest clearance, would be particularly short-lived in highly productive forests and factors such as species, climate, soil water and nutrient availability will operate to influence the productivity of the regenerating vegetation.

Unfortunately information about changes in hydrology following rainforest logging and subsequent phases of regeneration with secondary forest is lacking. Nevertheless, comparisons of annual evaporation totals from a study in secondary forest (Holscher *et al.*, 1997) and from investigations in primary forest reported by Shuttleworth (1988) indicate very little difference. Given

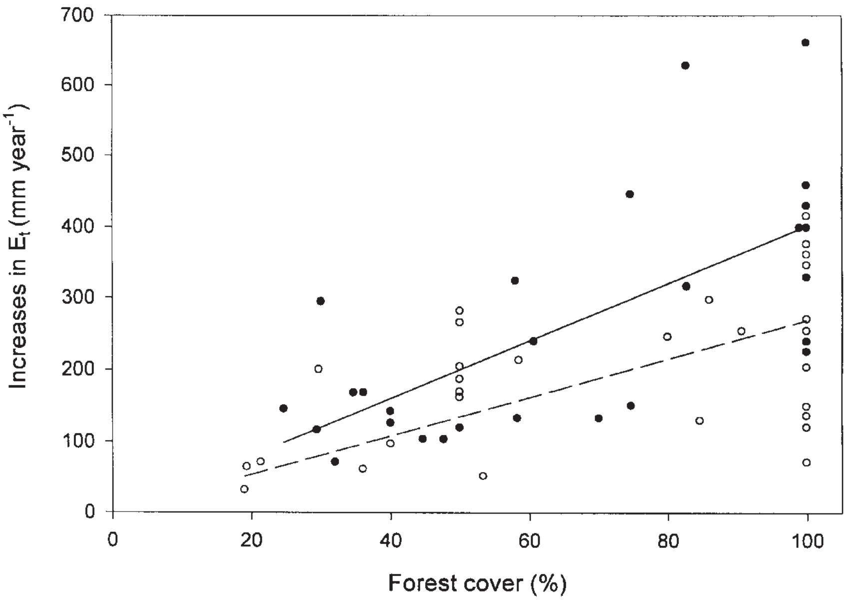

Fig. 15.4. Forest cover and evaporation (E_t) (——, conifers and eucalypts; ----, hardwoods; after Bosch and Hewlett, 1982).

the relevance of rainforest logging and its environmental impacts it is recommended that direct comparative studies of hydrology and controlling processes in primary and secondary rainforest are given priority. An additional priority is the hydrological impact of forest regrowth in abandoned pastures in the tropics.

Much of the catchment data from South Africa relates to afforestation of areas previously under native scrub and grasslands with pine and eucalypts in areas of the country with rainfall above 700 mm. Bosch (1979) found a reduction in streamflow by 35 mm for 10% forest increase. Van Wyk (1987) examined the impact of afforestation of three grassland catchments and found a decrease in streamflow ranging from 32 to 47 mm for 10% afforested. Dye (1996) examined data from a range of South African studies. The effect of afforestation on reducing streamflow occurred over a shorter period where the difference between evaporation from the original vegetation and mean annual precipitation was greatest. The rate of increase in evapotranspiration after afforestation was greater in eucalypts than it was for pines (Fig. 15.5). Dye (1996) listed a number of measures which might be employed to reduce the hydrological impact of forest plantations. Of course, some of the water saving that might follow from these suggestions would need to be balanced against implementation costs and loss of timber volume (value) increment.

1. Choice of a tree species with relatively low water use over the rotation. In South Africa, therefore, pines would be preferable to eucalypts;
2. Use of clones with low transpiration rates;

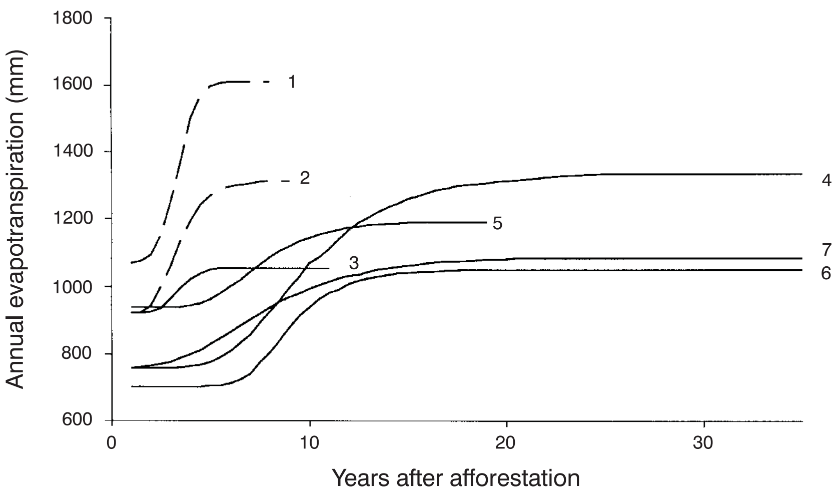

Fig. 15.5. Trends in evaporation following afforestation in seven paired catchment studies. 1. Westfalia D, *Eucalyptus grandis*; 2. Mokobulaan A, *E. grandis*; 3. Mokubalaan B, *Pinus patula*; 4. Cathedral Peak III, *P. patula*; 5. Lambrechtsbos B, *P. radiata*; 6. Bosboukloof, *P. radiata*; 7. Biesievlei, *P. radiata* (after Dye, 1996).

3. Development of a vegetation of low water use in riparian zones;
4. Control of understoreys in plantations;
5. Removal of plantations from catchments particularly those with a high runoff/rainfall ratio; and
6. Expansion of forests in low rainfall areas with low runoff/rainfall ratios.

Some of these recommendations might be of general application or specific to the South African situation. Nevertheless, recommendations such as number 5 might have to be considered carefully where the forest also has a flood prevention role.

Vertessy (1999) lists data from a large number of studies carried out in south-eastern Australia. Streamflows are generally less from forests than from grasslands because of the higher evaporation from the forest. Annual evaporation from grassland is generally less than 700 mm but for forests it can approach 1300 mm. Vertessy (1999) indicated that there was a considerable variation in evaporation from forests which is due to rainfall, species, productivity and age effects. Vertessy (1999) also summarizes a substantial amount of work from forest catchments in Australia. In three catchments in Victoria, Nandakumar and Mein (1993) examined the effect of clearance of eucalypt forest on streamflow which increased by 33 mm for 10% of forest removed. The magnitude is comparable to South African catchments with scrub afforested with pines or eucalypts (Dye, 1996). Cornish and Vertessy (2001) examined water yield increases in six catchments previously having a cover of eucalypt species in New South Wales, eastern Australia. In four catchments, streamflow increase was 40–50 mm per 10% of forest area removed. Another catchment showed a much bigger increase (70 mm) which was assumed to be associated with greater soil disturbance during logging. A sixth catchment did not show a streamflow increase. In this catchment, only a small area (25%) of forest was cleared and the effect was presumably close to the detection limit of such catchment experiments referred to by Bosch and Hewlett (1982). Apart from logging, additional associated activities such as burning and mechanical disturbance by vehicles particularly, can reduce soil infiltration and cause an increase in runoff (Cornish, 1991).

Studies in New Zealand have focused on the impacts on streamflow of converting either indigenous mixed evergreen forests or native tussock grassland to high production pine plantations. Particularly when replacing grassland, pine afforestation leads to a significant reduction in streamflow. Dons (1986) found an equivalent reduction of 30 mm in streamflow for a 10% increase in forest area. Fahey and Jackson (1997) indicate that streamflow increased by between 22 and 67 mm for 10% of native forests cleared in New Zealand's South Island; the average was 44 mm per 10% of forest cleared.

The major UK catchments used to investigate the effects of afforestation comprise a comparison of paired catchments in the headwaters of the Wye (upland pasture) and the Severn (forested, 70%). The mean differences in streamflow between these catchments based on data for 10 years from 1975 to

1985 was 198 mm (Kirby *et al.*, 1991) which, when based on the area of the Severn catchment afforested, equates to a reduction of streamflow of 28 mm per 10% of forest cover.

Bruijnzeel (1990, 1996) has reviewed the impact on streamflow of tropical deforestation. In the case of clearance of tropical forest, removal of at least 33% of forest cover resulted in significant increases in annual streamflow during the first 3 years, whereas initial gains in water yield after complete clearance ranged between 125 and 820 mm year^{-1}. The increases in water yield proved to be roughly proportional to the fraction of biomass removal and it is clear that changes in water yield mainly reflect the different evaporative characteristics of mature tropical forest and young secondary or planted vegetation. There is considerable variation in the first response to clearing that is only partially explained by differences in rainfall between the various study sites. Other factors include elevation (influencing evaporation), and catchment steepness and soil depth which will influence the residence time for water in the catchment. Also important is the impact of disturbance of the understorey and soil by machinery or fire (Malmer, 1992). The fertility of the soil will also influence post-clearing plant production and water use (Brown and Lugo, 1990). While maximum changes in streamflow after afforestation of grassland under South African conditions were in the range of 400–500 mm (Dye, 1996), recent studies by Waterloo (1994) in Fiji showed that the difference in water use between plantations of *Pinus caribaea* and *Pennisetum polystachyon* grassland was between 700 and 900 mm year^{-1}.

A major difficulty in assessing the impacts of afforestation and deforestation arises where surface water is largely absent, with recharge to groundwater dominating. Unless considerable afforestation/deforestation occurs the impact on groundwater levels may be even more difficult to detect statistically than is the case with small changes in forest cover on surface water catchments. Investigations into the effects of planting eucalypts on water resources in south India showed important impacts (Calder *et al.*, 1997). Deficits of soil moisture were greatest under eucalypts compared to other tropical tree species and crops. In the case of eucalypt stands, annual evaporation sometimes exceeded rainfall with the extra water coming from deeper and deeper exploitation of soil water by the eucalypt roots. However, given the relatively small area of forest planted in these semi-arid landscapes so far, it would be difficult to evaluate the impact of planting on groundwater levels, especially as other significant influences on groundwater levels (e.g. increase in bore hole abstraction for human use) have increased coincidentally. Therefore it is only from plot scale measurements, for example, of soil moisture abstraction, that differences in water use by forests can be compared with other trees and crops.

All the cases given above have shown that forest removal leads to greater or lesser increases in streamflow. There are, however, circumstances in which clearance of forest often leads to decreases in streamflow. As a ratio with incident rainfall, streamflow totals in tropical montane cloud forests are amongst the highest reported for any tropical forest (Bruijnzeel, 1990). The explanation of the situation is that a combination of extra (compared to short vegetation) input of occult precipitation to the forest from fog means that streamflow from forest areas can be significantly greater than from short vegetation. There tends to be a large amount of variation in fog interception in the different studies reviewed by Bruijnzeel and Proctor (1995) and so there is limited scope to draw quantitative conclusions on the effects of manipulation of tropical montane cloud forests on the decrease in streamflow.

Unlike the many cases cited above in which adding or removal of forests had substantial impacts on streamflow, there are, however, circumstances in which afforestation or deforestation may have little impact on annual water yield because evaporation differences between forest and alternative vegetation (e.g. grassland) are small. In lowland southern UK, an increase of 50% in the area of deciduous broadleaf woodland is being planned for by the middle of the next century. Preliminary studies by Harding *et al.* (1992) indicated that annual water use by broadleaf woodland is less or equal to grassland implying little impact on water resources of afforestation with broadleaves. Calder (1998) has, however, interpreted the Harding *et al.* (1992) data in a different way suggesting that broadleaf evaporation may exceed that of grassland by up to 38%. The Institute of Hydrology is currently engaged on a comprehensive and direct comparison of broadleaf and grassland evaporation to resolve this controversy. Because the types of broadleaf afforestation envisaged by the UK Government are on groundwater catchments, surface water measurements are not an option. As in the studies on eucalypts in south India referred to earlier, comparative soil water measurements are being used to measure evaporation from broadleaf woodland and grass.

2.3 Forestry and the timing of streamflow

As well as the effects of forestry activities on annual flows, catchment managers also require to know how the streamflow regime might change in response to forestry practices. From a sustainability point of view, the magnitude of low flows and maintenance of flow are probably as important as average annual water balance components. The reductions in low flow (in the case of afforestation) or increase in peak flows (in the case of forest clearance) can have particularly serious ecological consequences but there are other considerations such as the risk to culverts and bridges from flows higher than the design range.

In general, streamflow in small watersheds tends to have sharper peaks, higher storm levels, and shorter periods of sustained flow than it does in large watersheds (Likens and Bormann, 1995). However, it is difficult to predict the impact of reduction or increase of forest area on low flows. The greater interception and transpiration losses of forests are likely to lead to higher soil moisture deficits in the dry season which would likely be associated with low flows. On the other hand, the deeper infiltration of water into forest soils because of the enhanced permeability associated with the presence of deep roots may mean a prolonged dry season release of water from the deep horizons. Bruijnzeel (1990) believes that the enhanced dry season flows associated with forests are more to do with the soils than the trees *per se*.

A large majority of catchment studies have shown that low, median and high flows decrease following deforestation (Hewlett and Helvey, 1970; Burt and Swank, 1992; Schofield, 1996). It is not obvious, however, if low and high flows change in proportion with annual flows or if some part of the streamflow range is affected more than another. Research in South Africa (Bosch, 1979; Bosch and von Gadow, 1990; Smith and Scott, 1992, 1997) has shown that low flow reductions are more affected by afforestation than annual flows.

Bosch and von Gadow (1990) compared mean monthly streamflows for the Cathedral Peak catchment in South Africa, before and after afforestation of grasslands with pine. Absolute reductions in streamflow were greatest during the wet season but reductions in the forested catchment were proportionally greatest during the low flow periods when the grass is dormant. A smaller effect was observed when forests replaced indigenous scrub vegetation which unlike grass is evergreen and active during the low flow periods. Smith and Scott (1997) examined a substantial amount of streamflow data from catchments throughout South Africa. Their comparison focused on reductions in annual streamflow and low flows following afforestation of grasslands with pines and eucalypts. Annual and low flows were more marked for eucalypts than pines. Smith and Scott (1997) also compared results from ‘optimal’ sites, having deep soils and a subtropical climate, and ‘suboptimal’ sites with poor soils and cooler mountain climates. Flow reductions were far less prominent for both eucalypts and pines when suboptimal sites were afforested. For eucalypts, particularly, low flows were reduced relatively more than annual flows.

Vertessy (1999) assembled evidence about impacts of clearance and regeneration of moist eucalypt forests on streamflow regimes. One study (Haydon, 1993) showed mean annual streamflow increased by 290 mm but indicated that in some months the contribution to streamflow increased and reduced in others. Vertessy (1999) also refers to the work of Watson *et al.* (1999a) who studied streamflow relationships of the Maroondah basin in Victoria. In a high-rainfall (rainfall > 1600 mm) catchment, low, median and high streamflows increased uniformly after forest clearance, then declined together as regeneration commenced. However, in drier (rainfall < 1200 mm) catchments, low flows were more severely reduced than at median and high flows, especially in the later stages of regeneration. In the drier Karuah catchments in New South Wales, Australia, Cornish and Vertessy (2001) showed that flows of all ranges increased immediately following logging. High flows tended to increase most, especially in catchments that had thinner soils and the most disturbance. By the time vigorous regrowth forest was established in the catchment, all flows had returned to pre-treatment levels though low flows declined below pre-treatment levels in the catchments which experienced the greatest annual streamflow changes. However, no such low flow reductions were evident in the catchments with thin soils. Overall, Cornish and Vertessy (2001) attributed most annual streamflow changes in the Karuah catchments to changes in baseflows. Mackay and Cornish (1982) compared stream hydrographs for undisturbed control catchments and catchments which were burnt and logged. Peak flows and stormflow values increased following high intensity burning and further increased by logging of timber after burning. Mackay and Cornish (1982) suggest that these flow increases are due both to reductions in evaporation and also to lowered soil infiltration capacity following soil compaction by machinery.

Fahey and Jackson (1997) showed that the conversion of tussock grasslands to radiata pine plantation in the Glendhu catchment of the South Island of New Zealand produced uniform decreases in flood peaks over the whole range of streamflows. They compared the flood statistics of two catchments before and after one was converted from tussock grassland to pine plantation and showed the frequency distribution of mean flood peaks for four different storm size classes for a 3-year period before and after afforestation. Mean flood peaks for each storm size class were similar in both catchments prior to afforestation. After canopy closure following afforestation, mean flood peaks were reduced by around 60% in all four storm size classes in the forest catchment.

The influence of land cover on floods and low flows in the Plynlimon forest/upland grass comparison have been summarized by Kirby *et al.* (1991). Early work in the UK had indicated that forests reduce flooding but the only quantitative work had been done with mature forest. More recent research has shown that the link between flooding and land use is more complex, and the flood potential of a forested catchment varies through the crop cycle and with land management. In the high-rainfall areas in the north and west of the UK, drainage of anaerobic, peaty soils is an important prerequisite to successful tree establishment and stability in high winds. This situation would not be atypical of many other parts of northern Europe. The establishment and persistence of drains plays an important role in the flood responses of recently afforested land. After some years, with canopy closure and infilling of drains by vegetation the flood response reverts to equal or less than the original upland pasture. It is suggested that for small storms the effect of the more intensive forest drainage network, which tends to increase peak runoff and decrease response times, is more than balanced by the effect of interception by the forest canopy, which tends to reduce flood peaks, both by reducing total runoff and by delaying arrival of water at the channels. At very high flows, rainfall interception is no longer of any consequence, and also the artificial drainage network in the forest is no more intensive than the network of overland flow routes and natural pipes that become active in the grassland catchments. In the Plynlimon experiment, comparison of low flows on the sub-catchments in both the forest and grass catchments showed none of the differences were linked to vegetation cover but more related to the geology in a particular sub-catchment.

The long-term effects of forest drainage on hydrology of a forested area have been reported by Robinson (1998). He presents results from a 30-year study which followed hydrological changes of a moorland area, Coalburn, in northern England, after ploughing for drainage, tree planting and establishment up to canopy closure. An increase in flood peaks occurs after the establishment of forest drains which persists for the first 10 years or so of the forest cycle. Changes were also observed in the low flow behaviour following drainage and forest establishment. The increase in base flow is thought to be associated with the installation of deep drains which augment the natural drainage system of the original moorland. Robinson (1998) reports that there has been a decline in the base flow index since drainage but the rate of change means that a return to the pre-forest condition will not be achieved during the rotation period of the forest there at present.

Bruijnzeel (1990) addresses the conflicting situation for tropical rainforests in that greatly diminished dry season flows have been reported in the literature and are usually ascribed to deforestation (e.g. Myers, 1986). At first sight this would seem to conflict with the evidence presented elsewhere that increase in total annual water yield follows removal of tall vegetation. Also in most tropical small basin studies the bulk of this increase in flow was observed in the dry season or baseflow conditions (e.g. Gilmour, 1977; Edwards, 1979). The conflict can be resolved, however, taking into account the net effects of changes in evaporation and the different scope for infiltration of water into the soil associated with different land uses. After forest removal if infiltration has decreased to the extent that increased amounts of water leaving the area as stormflow exceeds the gain in baseflow associated with decreased vegetation evaporation, then a reduction in dry season flow will result. Reduced infiltration may result from the use of heavy machinery or by a substantial increase in the area of the catchment covered by impervious surfaces (e.g. roads and dwellings). It is clear from a review of hydrology of Tropical Montane Cloud Forest (Bruijnzeel and Proctor, 1995) that there is very little information about the impact of deforestation on annual flows and seasonal flows but the authors do suggest that seasonal flows may be substantially changed by modifications of soil infiltration during forest clearance.

2.4 Streamflow and forest age

One of the best examples of how catchment hydrological studies can be used to study the impact of land use change comes from the Melbourne area in south-eastern Australia. Long running catchment studies were already in place prior to serious bushfires in old-growth mountain ash (*Eucalyptus regnans*) forest (Fig. 15.6) which occurred in 1939. Figure 15.7 shows the sharp increase in

Fig. 15.6. Old growth forest of *Eucalyptus regnans* (alpine ash) forming part of an important water catchment for the city of Melbourne, Victoria. Such forests are not harvested, but are managed for water production, conservation values and recreation. Protection from wildfire is important because intense fires kill most trees and conversion to rapidly regenerating even-aged regrowth causes a significant reduction in water yield. Measures of old-growth habitat values, water yield and quality, and recreational value are important.

streamflow following the fire and subsequently the decrease in streamflow to levels less than those before the fire occurred. The interpretation of these results is that immediately after the fire there was no vegetation so evaporation was negligible but in a few years vigorous regrowth established a juvenile mountain ash forest evaporating at greater rates than the old-growth (Langford, 1976).

However, the year to year fluctuations suggest that catchment studies are not a short-term option if managers require to establish trends in which they

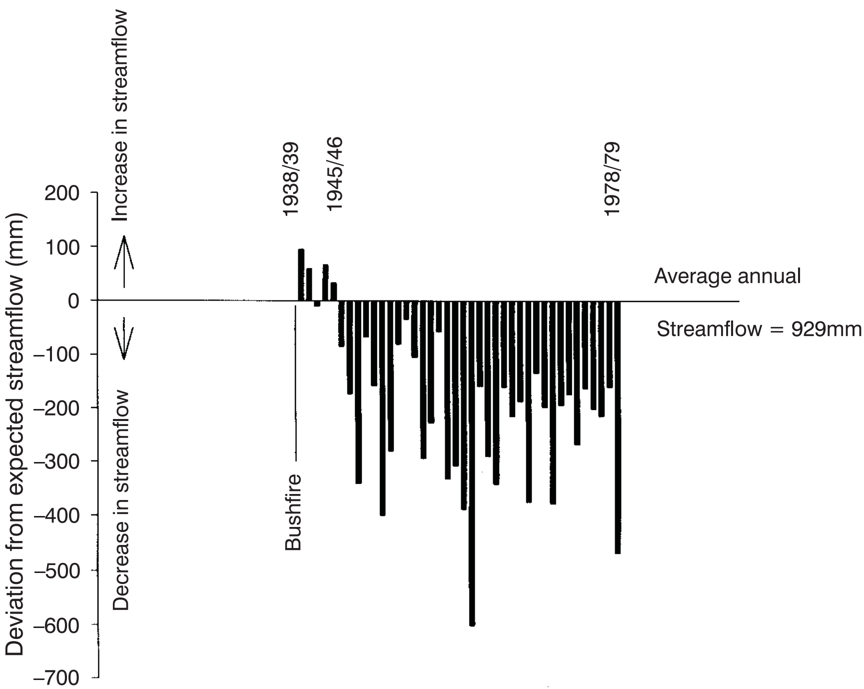

Fig. 15.7. Patterns of changes in annual streamflow from the Watts River catchment following the 1939 bushfire (after Melbourne and Metropolitan Board of Works, 1980).

have a high degree of statistical confidence. The fluctuation of water yield from catchments in the Melbourne area in relation to age of regrowth was further examined by Kuczera (1985, 1987). Water yield declines to a minimum 27 years after the fire and yield rises back to pre-fire levels by 200 years. Studies have also investigated the hydrological effects of clearing old-growth eucalypt forest and subsequent regeneration on the Karuah catchments in New South Wales (Cornish, 1993). He showed that water yields decline to levels significantly below pre-logging levels after about 6 years of growth, supporting the contention that evaporation of regrowth exceeded that from old-growth forest. Clearly substantial impacts such as forest clearance or catastrophic fires show a large effect on streamflow and a more gradual recovery if forest is allowed to re-establish. However some changes are more gradual and it may be several years before a statistically valid result emerges. Figure 15.8 shows the differences between rainfall and streamflow in two adjacent catchments in mid-Wales, UK (Hudson *et al.*, 1998). The forest catchment is of spruce planted around 50 years ago and is compared with an upland grass catchment. At the beginning of the study when the trees were around 30 years old there are large differences in streamflow implying higher evaporation losses by the forest.

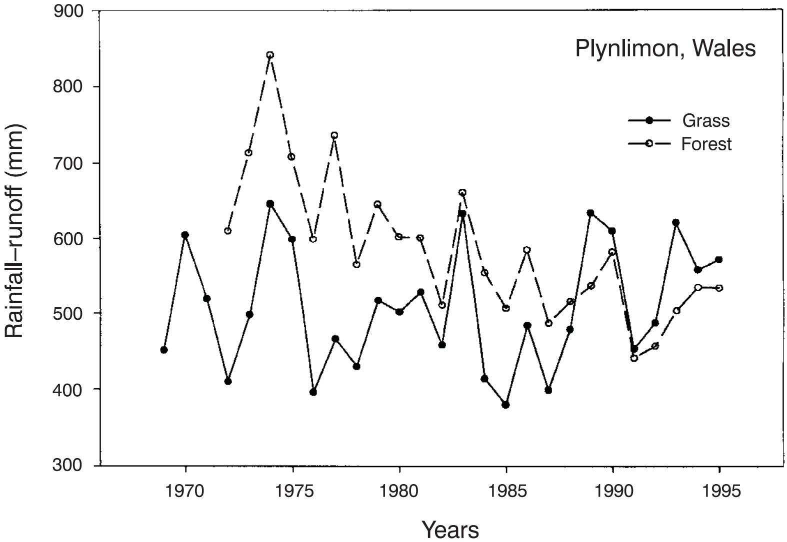

Fig. 15.8. Time trends of evaporation (precipitation – streamflow) for the forest (Severn) and grass (Wye) at Plynlimon, Wales (after Hudson *et al.*, 1998).

Detailed process studies at that time confirmed the higher evaporation of spruce with high rainfall interception losses being responsible (Calder, 1976). With the passage of time the differences in streamflow between the forest and the grass catchment have decreased. One explanation is that tree transpiration has declined with age and reduced the impact of the higher interception loss of the forest.

Some understanding of the mechanisms involved in the decline of transpiration of forests as they age is beginning to emerge. Modelling of data from mountain ash forests near Melbourne by Watson *et al.* (1999b) lead to the conclusion that declining leaf area and individual leaf transpiration efficiency account for increases in water yield after the initial high water use and low yield from the vigorous regrowth. These authors show that as forests age the ratio of sapwood area to leaf area declines. However, sapwood velocity is maintained with age in mountain ash (Vertessy *et al.*, 1997) suggesting that there is a reduction in leaf conductance with age as well as a reduction in leaf area index. The observation that regrowth has a higher water use than original forest has also been made by Hornbeck *et al.* (1997) who ascribe the higher water use by the regrowth to higher stomatal conductances of the colonizing species. Mencuccini and Grace (1996) examined the hydraulic properties of trees of Scots pine ranging from 7 to 60 years in Thetford Forest, East Anglia. They found that hydraulic conductance increased to a plateau at around 15–20 years and then declined. The authors calculate that the effects will reduce transpiration to about 70% of the rate in saplings.

One factor which might delay the full impact of afforestation on streamflow being observed is competition between the young trees and vegetation already growing on the site. One of the factors which Robinson *et al.* (1998) implicate in delayed establishment of full canopy cover at Coalburn is competition with heather (*Calluna vulgaris*) which dominated the original vegetation at the site. There may be low levels of transpiration and interception by the trees in young forests, because the trees do not fully occupy the ground space, or in old forests because the trees have crowns which have become moribund. However, at the stand level these effects may be compensated for by a substantial contribution from an undergrowth component in forests, if light levels permit their growth. The contribution that forest understoreys can make to the annual water balance at a site can be substantial. Black and Kelliher (1989) reviewed the literature about understorey transpiration in forests and demonstrated a number of cases in which 30–50% of the forest transpiration came from the understorey.

Key aspects of forest productivity (e.g. net primary productivity, wood production, photosynthesis and leaf area) have all been shown to increase as forests establish and grow, peaking at a particular age, which varies between species, and followed by decreases (Ryan *et al.*, 1997). It is fair to assume (Cornish and Vertessy, 2001) that stand water use fluctuations, which follow the growth stages of forests, might similarly follow the patterns observed in parameters used to describe aspects of forest production which change as a forest stand passes through the various growth stages from establishment through to eventual decline with age. For a given species, good relationships have been shown between leaf area per tree and diameter at breast height or sapwood area (Grier and Waring, 1974; Whitehead, 1978; Kaufmann and Troendle, 1981). Measurements of stand basal area or sapwood basal area can be used, in comparison with maxima expected for the particular species in a given region, to indicate how close to the peak growth rate is the growth of a particular stand. In a relative sense this comparison could serve as a valuable indicator of how water use is changing with age. A complicating factor in forest water use will be the compensatory role that might be served by forest understoreys. These might benefit from an increase in light penetration, because of reduced leaf area index in ageing forest stands, by increased growth and water use.

2.5 Modelling of forest water use

In catchments, topography, soil characteristics, vegetation and climate interact in a complex way to determine the nature and location of streamflow production and the associated transport of sediments, chemicals and organic debris through the landscape. Until recently, however, the endeavours to improve understanding of vegetation functioning and physical processes in soils, for example, have proceeded somewhat separately from catchment hydrology studies. Catchment studies, of which many examples have been referred to above, have been required to answer questions such as; what will be the effect on streamflow of a change from grassland to eucalypts? The mechanisms by which differences arise are of secondary importance. The development of robust management strategies, which might include indicators for sustainable development, should be based on a detailed understanding of processes. It would be risky to extrapolate conclusions from catchment studies without a better knowledge of mechanisms resulting in the catchment behaviour.

A particular problem with many catchment studies and one which process forest hydrologists find so frustrating is the lack of information about the relative contribution of rainfall interception losses and transpiration to the evaporation losses from the vegetation. For South Africa, Dye (1996) provides useful references to interception studies located close to some of the catchment studies he also reports. Interception losses from *Eucalyptus grandis* and *Pinus patula* are 4.1% and 13%, respectively. These low values reflect the infrequent, intense storm pattern occurring predominantly in the summer-rainfall region. From these low values it is possible to appreciate that in the South African catchment studies the transpiration process is the dominant mechanism by which high water losses from canopies occur, and that differences in soil water uptake go a long way to explain differences between the water use of various species. In complete contrast, studies under the very different rainfall conditions of the UK have shown (e.g. Calder, 1976) that differences in transpiration between conifers and short vegetation make very little contribution to the water use differences between forest and grass, but which are explained by the high rainfall interception losses particularly because rainfall arrives predominantly in long-duration, low-intensity storms.

In the case of evaporation of intercepted rainfall we have available a range of models which have been validated for a wide span of forest types ranging from conifers and broadleaves in northern and southern temperate regions, tropical rainforest and sparse forests (Rutter *et al*., 1971, 1975; Gash, 1979; Calder, 1987; Lloyd *et al*., 1988; Teklehaimanot *et al*., 1991; Gash *et al*., 1995; Ubarana, 1996). For transpiration losses from forests there is now a substantial amount of information from a wide range of forests although there is still a requirement for more information for tropical forests and semiarid woodlands. A common approach to modelling and predicting forest transpiration is to use the Monteith–Penman formula (Monteith, 1965). This enables transpiration to be calculated from meteorological data with surface conductances measured micrometeorologically (Shuttleworth, 1989) or stomatal conductances measured at the leaf level (Roberts *et al*., 1993). Often submodels are used to estimate surface or stomatal conductance from weather data (e.g. Jarvis, 1976). There would be considerable merit in testing how widely some of the surface conductance functions already published can be extrapolated to other forests. Unfortunately, there are a number of features of the detailed forest micrometeorology and ecophysiology studies which have generated the core data which may limit their usefulness, at least to some degree; studies have usually been confined to mature stands, growing in flat areas and often on soils which are not flooded. Therefore there are, for example in plantation forests, unresolved questions about how transpiration changes as forests establish and mature. We know very little about transpiration of forests on thin soils or those growing in flooded areas. Equally, scientists have tended to locate experiments in good or elite stands and avoid low grade sites and this selection may have biased results also.

The detailed understanding and predictive ability that has come from studies of the type described above has now been encapsulated as key elements in soil–vegetation–atmosphere schemes such as BATS (Dickinson *et al.*, 1986) and SiB (Sellers *et al.*, 1986). However, from a hydrological point of view these schemes fall short because there are no attempts to redistribute moisture laterally. In contrast, traditional hydrological models have been concerned mostly with runoff production; the description of vegetation functioning is often a stipulation of potential evapotranspiration associated with a simplified soil moisture stress relationship.

In recent years attempts have been made to bring together hydrology and vegetation behaviour in distributed hydrological models of catchments. There are a number of such models, e.g. SHE (Abbott *et al.*, 1986a,b), SHETRAN (Dunn and Mackay, 1995). The Regional Hydroecological Simulation System (RHESSys; Band *et al.*, 1993) is a particularly comprehensive model and has predicted daily and total annual runoff particularly closely compared to measured values (White and Running, 1994). These models aim to be physically and physiologically realistic and use digital terrain information to provide topographic responses using models specific for the purpose, e.g. TOPMODEL (Beven and Kirkby, 1979; Beven, 1997); TOPOG (Vertessy *et al.*, 1996). Wigamosta *et al.* (1994) describe the Distributed Hydrology Soil Vegetation Model (DHSVM) in which downslope redistribution of soil moisture via saturated subsurface flow is explicitly modelled on a pixel by pixel basis. Other distributed hydrological models have tended to use distribution of soil moisture on a statistical basis e.g. TOPMODEL. A key focus of DHSVM (Wigamosta *et al.*, 1994) is an attempt to model the fate of snow in catchments in which precipitation in this form dominates. Watson *et al.* (1999b) used a distributed hydrology model (MACAQUE) to emulate streamflow over extended periods (~100 years) for large catchment areas in the Maroondah catchments in Victoria, Australia. These authors emphasize the importance that rainfall values used in the model have in the prediction.

The limitations for the distributed modelling approach and the choice of a particular model may well be determined by the availability of data and parameterization for the locality rather than physical realism and understanding in the model. Although, there are still some processes where further understanding is required. The lack of local data is likely to be most acute in remote tropical regions. Key data will be climate, especially rainfall of adequate frequency, evaporation characteristics of the vegetation, hydraulic properties of the soil, channel geography and topographic maps. Of these features, however, it would be most important to know most about the parameters which are changed during any forest management. The parameters principally affected would be the evaporation characteristics of the vegetation and soil hydraulic properties.

In the event of insufficient data being available to run distributed hydrological models an alternative is to examine forest water use and its components derived from appropriate process studies in the relevant region. This approach is pragmatic, largely empirical and thus far can only be used where substantial data sets enables and warrant generalization. This will be the case for forests and woodlands in north-west Europe but perhaps also for tropical forest. Roberts *et al.* (1998) have assembled data on transpiration and interception losses from a wide range of forests from which it is possible to infer similarities and differences.

On an annual basis conifers exhibit a higher interception loss as a percentage of annual rainfall (interception ratio) than broadleaf trees (Fig. 15.9). In western European conditions rainfall interception as a percentage of gross rainfall by conifers is normally between 25 and 35% and around 15–25% by hardwoods. In contrast to the differences in interception loss, the margin between conifers and hardwoods in transpiration (Table 15.1) is small (Roberts, 1999).

The annual transpiration of both types (~300 mm) is far less than expected from potential evaporation calculations using the Penman equation

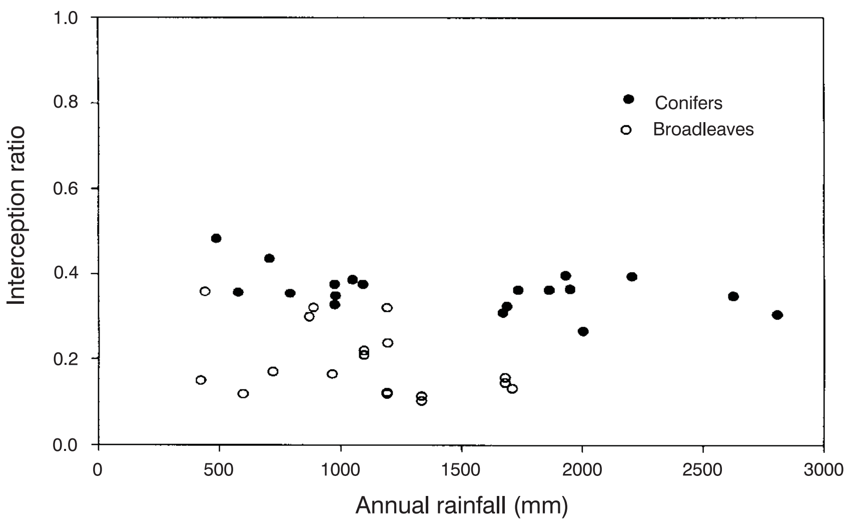

Fig. 15.9. Interception ratios plotted against annual rainfall for interception studies involving conifers and broadleaves in Europe (after Roberts *et al.*, 1998).

(Penman, 1948). In contrast annual transpiration by short rotation coppice willow and poplar is about 50% larger than conifers and broadleaf trees. Short rotation poplar coppice does not show the feedback responses of stomatal conductance to air humidity deficit and very high daily transpiration rates are often observed. However, transpiration is often restricted eventually by the onset of soil water deficits (Hall *et al.*, 1996) a situation which occurs much less often in conifer and broadleaf plantations. It is interesting to make a rough calculation of evaporation losses for temperate broadleaf woodland based on annual transpiration of 300 mm year^{-1} and an interception loss of 20% of annual rainfall, say 1000 mm, giving a total of 500 mm. In fact, 500 mm year^{-1} evaporation was the stable annual total observed by Likens and Bormann (1995) and illustrated in Fig. 15.2.

Table 15.2 compares the relative contributions to annual total evaporation from a tropical and a temperate forest. Annual transpiration from tropical forest is three times that of the conifer forest but of course there are few seasonal limitations in the Amazon. It is notable that because the relationship of g_s and air humidity deficit is similar, and surface conductances are comparable, *daily* transpiration rates for tropical and temperate forests are rather similar, around 3.5 mm day^{-1} (Shuttleworth, 1989). Studies in the Amazon forest showed that transpiration was very little affected by the observed reductions in soil moisture content.

The lower percentage interception loss for tropical forest is a consequence of the different rainfall conditions; the rainfall in the Amazon is dominated by convective storms which are short and intense, in contrast to the predominantly frontal rain in western European conditions (Thetford) with long-duration, low-intensity rainfall. Under these conditions the forest canopy remains wet for long periods. This enables substantially more evaporation from the canopy to occur especially as evaporation can occur during the rainstorms if the air is not saturated. In some cases high interception losses,

Table 15.1. Annual transpiration of temperate tree types (after Roberts, 1999).

Conifers	338 mm ± 43 (n = 9)
Hardwoods	302 mm ± 59 (n = 19)
Short rotation coppice	457 mm ± 28 (n = 4)

Table 15.2. Water balance components (mm) for Thetford Forest, UK (1975) and Reserva Ducke, Brazil (1984).

	Rainfall	Interception	Transpiration	Evaporation	Drainage
Thetford	595	213	352	565	30
Ducke	2593	363	1030	1393	1200

Data from Gash and Stewart (1977) and Shuttleworth (1988).

as a percentage of gross rainfall, have been recorded from tropical forests. Schellekens *et al.* (2000) found interception losses of up to 40% in a rainforest in Puerto Rico. The explanation of the high losses was the frequent small rain showers and energy provided by advection from the nearby Atlantic Ocean. This situation parallels the high forest interception losses for the UK for which a similar explanation is offered. Water use by fast-growing species in tropical climates may be considerable. In a study in South India, Roberts and Rosier (1993) showed that transpiration could be as high as 6 mm day^{-1}, when adequate soil water was available, falling to less than 1 mm day^{-1} when soil water was limiting.

The link between high transpiration rates and soil moisture deficits in fast-growing trees (short rotation coppice and eucalypts) is relevant to Rutter's (1968) similar observations that forests with high rates of water use showed a response to soil water deficits over a wide range of deficit (Fig. 15.10a) while trees with low transpiration rates were not so restricted (Fig. 15.10b).

This 'rule of thumb' analysis suggests that temperate broadleaves and conifers will transpire about 300 mm $year^{-1}$ with conifer interception losses being around 30–40% of gross rainfall, double that of broadleaves. Transpiration will be little affected by soil moisture deficits. Fast growing coppice will transpire 50% more than traditional broadleaves and conifers but interception loss will be similar to traditional broadleaves. Short rotation coppice transpiration might be limited by soil moisture deficits. Tropical rainforest will transpire around 3.5 mm day^{-1} and interception loss will be between 10% and 15% of gross rainfall. Soil moisture deficits will not restrict transpiration. In contrast, fast-growing tropical tree plantations can transpire up to 6 mm day^{-1} and can be restricted by soil moisture deficits. Interception losses will be similar to tropical rainforest and will be low where rainfall comes in short intense storms. Interception losses of tropical forests will increase if rainfall is distributed as many light showers. What is not included in this simple approach is how transpiration and interception of the different forests will fluctuate in response to forest age, a factor which was identified earlier in this chapter as being of particular importance. This section has attempted to show that distributed models may be an ideal to strive for, but that far less sophisticated approaches can reveal important trends which can serve as predictions. To take this approach forward requires a synthesis of forest water use data for regions other than the north temperate zone.

3 Forests and water quality

3.1 Forests and stream sediment

In undisturbed forest catchments, patterns of sediment movement are very different from that of dissolved material (Likens and Bormann, 1995).

Whereas the concentration of dissolved material does not change with flow regime, sediment yield increases almost exponentially with flow, a large percentage of the sediment discharge being associated with the highest flows.

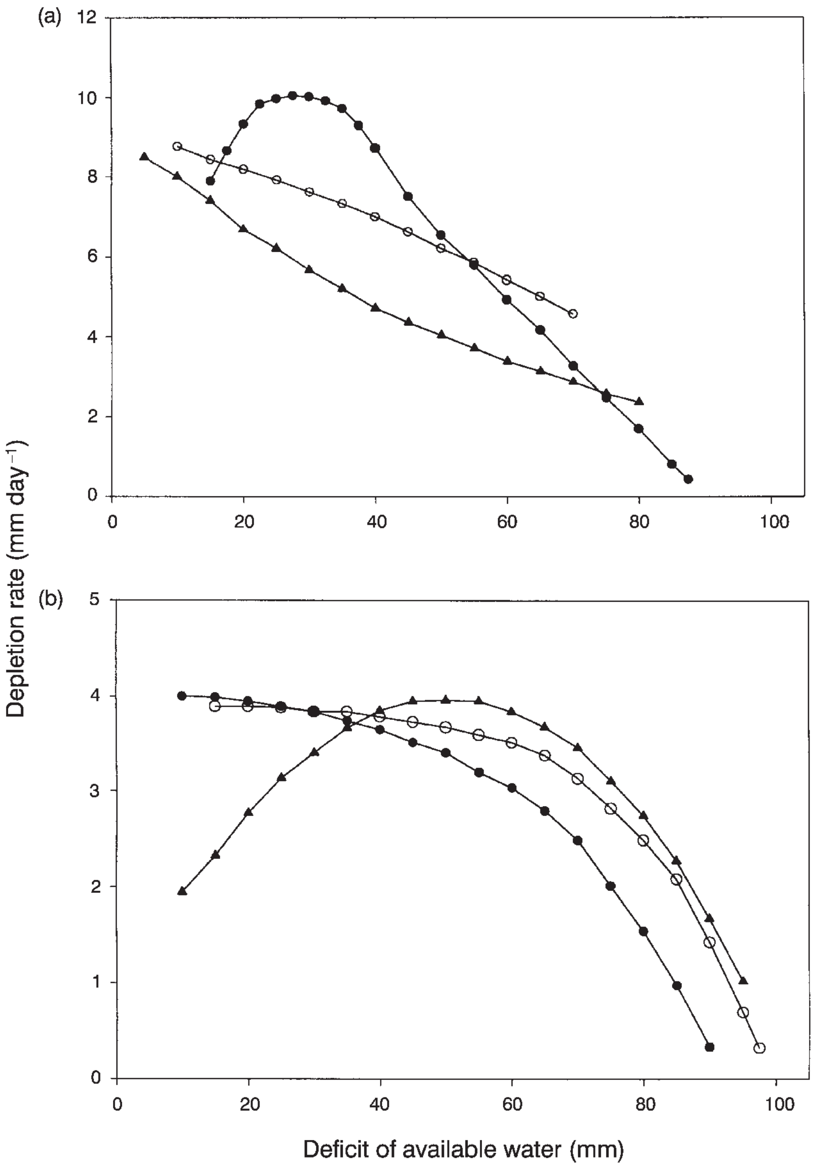

Fig. 15.10. The relation between the rate of soil water depletion and the deficit of available water under forests: (a) △, Moyle and Zahner (1954); ●, Metz and Douglass (1959); ○, Zahner (1955); (b) △, Croft and Monninger (1953); ●, Rowe (1948); ○, Zinke (1959) (redrawn after Rutter, 1968).

The nature of the sediment also changes with flow regime. Under low flow conditions the sediment is dominated by organic material derived from fresh and decayed plant material and humus. At higher flow regimes, inorganic material from the soil and recently weathered from rocks predominates. Particularly at high flow regimes, the sediment yield may also include material, previously deposited in the stream channels, remobilized by the higher flows.

Sediment is composed of two fractions, suspended sediment, which is fine particles of soil and debris in suspension in the stream water, and bed load, comprising larger-sized material washed along the stream bed. Forestry activities cause a substantial increase in both components of sediment and both can contribute considerably to instream and downstream impacts of forestry activities. Sediment accumulation in streams, rivers, lakes and reservoirs can cause flooding, limit the potential for navigation and reduce the storage capacity of reservoirs. The major effects on stream biota will come from the suspended sediments and possibly smaller fractions of bed load. Suspended sediments will increase stream turbidity and seriously reduce the light levels for both plant and animal stream dwellers. Fine sediments depositing on the stream bed can reduce the potential sites for egg deposition by fish. The export of some chemicals, particularly phosphorus, is allied to sediment movement, because the chemical tends to be bonded strongly to soil particles, unlike most other nutrients which are leached out of the soil in solution.

Soil erosion is a natural geomorphic process which is significantly accelerated by human action. It is this acceleration of the natural erosion processes which is the issue for SFM. Various aspects of forest harvesting all contribute substantially to levels of sediment production, far in excess of background levels observed in undisturbed forests. These activities include road construction, operations at logging sites, use of logging trails and removal of logs by road. Two mechanisms contribute to sediment production from land disturbed by forestry activities. Removal of the vegetation cover, trees, understorey and forest floor litter expose the soil surface to direct contact with raindrops and detachment of soil particles and soil compaction follows. Compaction of the soil surface at harvest sites, log landings, skid trails and roads is also substantial. Runoff will be enhanced from these compacted areas, with associated entrainment of soil particles. Surface runoff, with which erosion would be associated, is very rare in undisturbed forests. However, it has been observed to occur under very intense rainfall, particularly where shallow impeding layers (or bedrock) below the soil surface prevent deep infiltration (e.g. Wierda *et al*., 1989).

There is a substantial volume of literature linking sediment yields with logging activities and roads (Reid, 1993). Emphasizing the importance of roads to sustainable forestry, Douglas (1999) states ‘roads are so much the focal point of erosion and sedimentation problems that they offer the greatest opportunity for making selective logging more sustainable’. Overall, studies seem to show a two- to 50-fold increase in sediment yields with a large number of cases being associated with badly located or maintained roads (Binkley and Brown, 1993). Reid and Dunne (1984) emphasize, particularly, the loss of small size (< 2 mm) particles from road surfaces; fine grained material which because of its size is most threatening to fish and water quality.

Croke (1999) gives a good summary of the impact of forest harvesting on water quality, particularly sediment production. The current state of knowledge on the effects of harvesting methods on water quality has emerged from two types of approach which have been used to study the rates of erosion and controlling mechanisms. Firstly, attempts have been made to make measurements of sediment production at the stream outlet either in a paired catchment study, with a comparison of a treated catchment with a control, or in a single catchment measured before or after perturbation. A second strategy has been an attempt to measure erosion rates on individual representative elements of the landscape (e.g. roads, tracks, log landings, harvest areas and undisturbed forest) followed by attempts to scale the measurements to estimate changes in catchment water quality resulting from forestry practices. Some of the problems with the first approach are its inability to identify the source of the sediments and determine if they result directly from forest logging or a consequence of sediment stored previously in the stream system being remobilized because of a changed stream hydrograph, caused by the logging activities. In addition, there are technical problems such as scaling from a few sampling events and differences in streamflow regime because of differences in rainfall in the measurement periods before and after logging. Croke (1999) expressed concern that very few long-term studies are available to assess the effects of forestry activities on water quality. In Australia, for example, only one study has a 30-year record of streamflow quality. This situation is similar in other developed countries but in developing countries the length of records is usually even less. Douglas (1999) also expresses concern that few sedimentological studies in forests extend significantly into the post-logging phases.

Short-term catchment monitoring studies have limited value in revealing the magnitude of the effect on sediment of a forest disturbance and identifying suitable options for remedial or preventative management for the catchment (Croke, 1999). The second, plot, approach to studying erosion has problems because of the scale of sampling. Not all of the eroded sediment will travel directly into the stream or reach the stream outlet of the catchment. The residence time for sediments within a catchment can be considerable and will have implications to monitoring programmes. There can be a big difference between the amount of sediment produced at any particular place, as a consequence of some forestry operation, and the amount delivered to a receiving stream.

Nevertheless, advances have been made recently in the implementation of both approaches referred to above. It is now possible to use tracers to identify specific sources of sediment and assemble sediment budgets for catchments (e.g. Walling *et al.*, 1999). Larger scale plot studies have been accomplished and these have enabled a more accurate assessment of contributions from particular landscape elements (e.g. Croke *et al.*, 1999). There has also been a substantial increase in our understanding of the individual processes involved. Rose (1993), for example, gives a comprehensive description of physical processes involved in erosion. Two recent studies (Croke *et al.*, 1999; Douglas, 1999) have discussed measures to prevent runoff from logging areas and tracks from reaching forest streams.

Croke (1999) identifies three topics which require knowledge and understanding to quantify the effects on water quality of forestry activities.

1. Information on sediment sources and their spatial distribution in relation to streams. Because different parts of a catchment contribute sediment unequally it is necessary to know the relative contribution from different sediment sources and their relationship to the streams.
2. There is considerable potential for sediment storage anywhere between the source and the stream. Knowledge of potential storage locations will allow identification of the delivery routes which pose the biggest threats to stream quality.
3. There is a requirement to assess the usefulness of the best management practices in relation to sediment production and delivery.

Two main agencies of erosion operate in relation to forestry activities, soil particle detachment by raindrops and detachment and movement of soil by surface water flow. Generally, rainsplash erosion is a consequence of reduction or removal of the tree and understorey canopy and litter layers which protect the soil from raindrop impact. However, there are cases, for example, tropical forest plantations without undergrowth, in which there is no barrier against the impact of raindrops as they fall from the canopy, and rainsplash erosion is likely. In fact, the leaves of some tropical trees modify the raindrop spectrum and the kinetic energy, and therefore erosion potential, of raindrops falling from leaves such as teak can be substantially more than the rainfall (Hall and Calder, 1993).

The mechanical techniques used in modern forest logging operations result in dense track and haulage road networks and large log landings – areas used for assembling logs for transport, debarking and loading. In total these can occupy as much as 16% of Malaysian logging sites and up to 25% in Australia (Douglas, 1999). Areas bare of vegetation, including roads and tracks become heavily compacted by movement of logs and vehicles. Also, without full vegetation cover, the energy of raindrops compact the soil surface layers and overland flow removes fine particles. Van der Plas and Bruijnzeel (1993) quote figures for infiltration at the Danum Valley, Sabah as 88 mm h^{-1} for undisturbed forest, 73 mm h^{-1} for regenerating forest but only 15 mm h^{-1} for logging tracks that were abandoned as long ago as 12 years. From studies in Palawan, The Philippines, Dixon (1990) estimated that although roads only accounted for 3% of the total forest area, they were responsible for 80% of the total erosion. Douglas (1999) referring to plot studies on abandoned logging tracks in the Danum Valley indicated that 52% of the rainfall becomes overland flow on these tracks compared to 5% under natural forests.

Croke *et al.* (1999) examined surface runoff from snig tracks and general harvesting areas. The highly compacted snig tracks were the dominant source of runoff exceeding that from harvesting areas by an order of magnitude. The relative differences in runoff production declined during extreme rainfall events. This effect has also been observed in larger-scale studies where the specific impact of compacted areas on the catchment hydrograph was only observed in small storms. Substantial recovery of infiltration of both harvest areas and snig tracks occurred after 5 years which was not related to bulk density changes. This rate of recovery is considerably quicker than other studies which have used compaction as a measure of recovery, but is in agreement with other runoff studies on disturbed forest sites elsewhere (Thurow *et al.*, 1993).

Croke *et al.* (1999) also measured saturated hydraulic conductivity (K_{sat}) on harvest areas and snig tracks. The harvest areas showed a more heterogeneous pattern of K_{sat} than the skidding tracks. K_{sat} of recent harvest areas was ~58 and ~12.5 mm h^{-1} for snig tracks. However, these values are much higher than those reported for unsealed forest roads by a number of authors (e.g. Reid and Dunne, 1984, ~0.8 mm h^{-1}; Luce and Cundy, 1994, ~0.11 mm h^{-1}; Malmer and Grip, 1990, ~0.28 mm h^{-1} and Ziegler and Giambelluca, 1997, ~1.15 mm h^{-1}). Ziegler and Giambelluca (1997) used a disc permeameter to measure saturated hydraulic conductivity (K_{sat}) on different land surfaces in northern Thailand. K_{sat} of unpaved roads was an order of magnitude lower than other surface types. K_{sat} rates were not exceeded by any recorded rainfall except on roads or roadside margins. Although roads occupy a small areal extent (0.5%) of land surfaces in the area studied by Ziegler and Giambelluca (1997), they contribute a large portion of runoff during frequently occurring, small rainfall events. During larger rainfall events, areas under agriculture, secondary vegetation and forest assume a greater importance to surface runoff because of their larger areal extent.

Croke *et al.* (1999) considered the effectiveness of cross banks, constructed after logging across snig tracks, in directing runoff into adjacent areas. Cross banks are approximately 0.5 m high and extend across the width of the snig track, they are designed to pond water flowing down the snig track and dissipate it into the adjacent hillside. Croke *et al.* (1999) suggest that, for 1–10-year extreme rain events, snig tracks would not contribute to surface runoff or sediment to the streams because of adequate dispersion of runoff into the forest. With more extreme events, however, the decreased effectiveness of hillslope infiltration at outlet points raises the issue about the distance snig tracks should terminate near to streams or buffer zones. Croke *et al.* (1999) believe that the primary objective of management should be to reduce the volume of runoff discharges at each cross bank outlet and especially on snig tracks close to the stream side or buffer. Cross banks can be used to delimit the upslope contributing area to each snig track element, and may be constructed at more frequent intervals along the snig track to eliminate excessive runoff volumes at the outlets of cross bars.

A major issue is the effective closure of logging and snig tracks after cessation of logging. Simply bulldozing earth barriers across the tracks and breaking up the compacted surface to allow infiltration and encourage regeneration of vegetation can reduce runoff and erosion considerably (Douglas, 1999). In studies in eastern Sabah referred to by Douglas (1999) water bars (presumably equivalent to 'cross banks' in the terminology of Croke *et al.*, 1999) were used in attempts to reduce the length of continuous flow and associated erosion on abandoned snig tracks. These water bars were ~1 m high and were spaced every 20 m along the track. The lower ends of snig tracks closest to stream channels are the earliest to revegetate presumably because they had been used less. An improvement to the management suggested by Douglas was to cover the water bars with vegetation trash to prevent rainsplash damage. Main log haulage tracks probably require more extensive treatment than the water bars proposed for snig tracks (Douglas, 1999). Douglas (1999) proposed breaking up of the surfaces of unused roads to promote vegetation regeneration and excavation of cross drains. Even after logging some unpaved roads will remain and will be in use by vehicles. Differences in runoff from roads which are in use and those abandoned have been shown by Reid and Dunne (1984). These authors also made measurements of sediment discharge on an unpaved forest road during rainstorms and identified substantial contributions of sediment associated with the passage of vehicles across the relevant road section during the storm. Douglas (1999) refers to regulations from Sabah which prohibit the use of some unpaved forest roads in the wet season.

Recent work by Luce and Black (1999) further examined aspects of roads contributing to sediment production. Most road segments produced little sediment, while only a few segments contributed a large amount. This information suggests that the most efficient management for sediment would be to concentrate on the road segments producing the most sediment. Sediment production was proportional to the product of road segment length and the square of the slope. This emphasizes that slope is an important aspect to consider for sediment budgets. Fine grain soils produce more sediment than coarser soils. The work by Luce and Black (1999) confirmed studies by others (e.g. Reid and Dunne, 1984), showing that older roads with undisturbed ditches yield less sediment than older roads with disturbed ditches.

3.2 Dissolved substances

Nutrient losses from forest lands not subject to disturbance are primarily determined by the nature of the geological substrate and the degree to which this has been weathered. Bruijnzeel (1990) reviewed the literature on the nutrient outflows in drainage water in a wide range of tropical forest types. Although there is a great deal of variation, Bruijnzeel (1990) was able to categorize nutrient release from forests on the basis of the soil quality on which the forests were growing. Apart from phosphorus, there were considerably greater quantities of nutrients in the water from forests growing on good soils than on poor soils (Table 15.3). Likens and Bormann (1995) make an important point about the need for frequent sampling of dissolved substances because of the impact of isolated events in which there are high concentrations of substances.

Generally speaking, the more infertile the soil the more 'closed' the nutrient cycle. Additionally, the more changes in land use are likely to upset the balance between nutrient inputs and outputs maintained by the forest nutrient cycle (Bruijnzeel and Critchley, 1994). Different land uses produce drainage waters of very widely differing constitution. Table 15.4 compares the losses and concentrations of nitrogen, phosphorus and potassium from a range of forest types and agricultural uses.

For a given catchment the gross output of chemicals varies closely with the annual streamflow. Figure 15.11 shows this relationship for four cations at the Hubbard Brook Experimental Forest (Likens and Bormann, 1995). The closeness of this relationship will serve well as a predictor of outputs against which to evaluate the effects of different treatments but Likens and Bormann (1995) emphasize the dangers of using the relationship off the catchment where it was obtained. These authors also warn against relying on infrequent sampling to draw conclusions about catchment behaviour. It is sometimes suggested that a few streamwater samples be collected to serve as a biogeochemical baseline for a terrestrial ecosystem. However, the marked and often unexplainable changes in an intensively sampled stream at Hubbard Brook suggest caution in attributing baseline characteristics to a few samples that unwittingly may characterize a high or low period in the history of the site. Clearly, the most useful biogeochemical baseline is one of sufficient length to allow trend analysis.

Old-growth temperate and tropical forests are dynamic ecosystems in which tree death is approximately balanced by growth. Uhl *et al.* (1988) estimated that 5% of tropical forest may be in a gap phase at any one time. Workers cited by Bruijnzeel (1990) showed that nutrients released by small gaps (up to 200 m^2) did not produce extra leakage of nutrients from the root zone. However, Parker (1985) working with larger gaps up to 2500 m^2

Table 15.3. Average nutrient losses (kg ha^{-1} $year^{-1}$) in drainage waters from tropical forests (after Bruijnzeel, 1990).

Fertility	Ca	Mg	K	P	N
Low–moderate	12.3	5.25	11.6	5.76	1.1
Moderate–high	222	38.4	23.2	1.47	18.8

showed that there was leakage to the soil but none reached streams. Kinniburgh and Trafford (1996) showed increased concentrations of nitrate in deep soil water below tree fall gaps in a beech forest in the UK; the beech forest had no understorey. In an adjacent ash forest with a vigorous understorey, nitrate was less prevalent in the deep pore water. The authors speculate that the extensive root mat of the understorey and ash trees serves to retain nutrients more effectively in the root zone.

In contrast to the low impact of tree fall gaps on streamwater chemistry, more widespread forest clearance produces a substantial release of nutrients

Table 15.4. Effects of different land uses on water quality: losses and concentrations of nitrogen (N), phosphorus (P) and potassium (K) (after Newton *et al.*, 1990).

	N	P	K
	Losses (kg ha^{-1} $year^{-1}$)		
Forest			
Slash and loblolly pine, USA	0.32	0.04	–
Deciduous hardwood, USA	2.0	–	–
Mixed coniferous, Canada	1.7	0.0	–
Native evergreen, New Zealand	1.45	0.12	–
Eucalypt, Australia	0.12	0.004	–
Spruce, UK	16.1	–	5.7
Aspen, USA	0.3	0.1	4.4
Evergreen oak, Spain	0.03	0.01	0.3
Agriculture			
Arable land, UK	4–13	0.06	–
Pasture, UK	8	0.05	–
Various intensive, USA	2–38	0.2–1.2	–
Pasture, USA	2–12	0.1–4.6	–
Maize, USA	2–62	0.1–2.5	0.4–26
	Concentration in runoff (mg dm^{-3})		
Forest			
Slash and loblolly pine, USA	0.08	0.01	–
Deciduous hardwood, USA	1.47	0.008	–
Mixed coniferous, Canada	0.23	0.01	–
Eucalypt, Australia	0.01	0.003	–
Rainforest, Amazon	0.004	0.01	0.15
Pine plantation, Florida	0.01	0.00	–
Agriculture			
Grassland, UK	4.1	0.09	–
Arable, UK	9	0.02–1.7	–
Various intensive, USA	5–25	0.13–0.33	4.0–11

from the cleared area. There will be release of material from decomposing vegetation but also the lowered evaporation after vegetation removal increases soil moisture content and drainage. Figure 15.12 shows peaks of cations and suspended sediment produced after clearfelling at Hubbard Brook (Likens *et al.*, 1978). However, after 2–3 years, levels of cations in the stream water returned to background levels. As expected, burning of forest residues after logging contribute substantially to increases in chemical loading to streams. As well as disturbances due to forestry activities, there are other circumstances which can cause pulses of nutrients to be released, possibly to enter streams: melting of snow packs, freeze/thaw cycles in the soil and soil rewetting after drought (Foster and Walling, 1978).

3.3 Stream salinity

Western Australia offers a good example of how injudicious removal of forest cover has lead to salinization of soil and streams. The source of excess soluble salts in the Western Australian soils is salt blown inland and deposited with

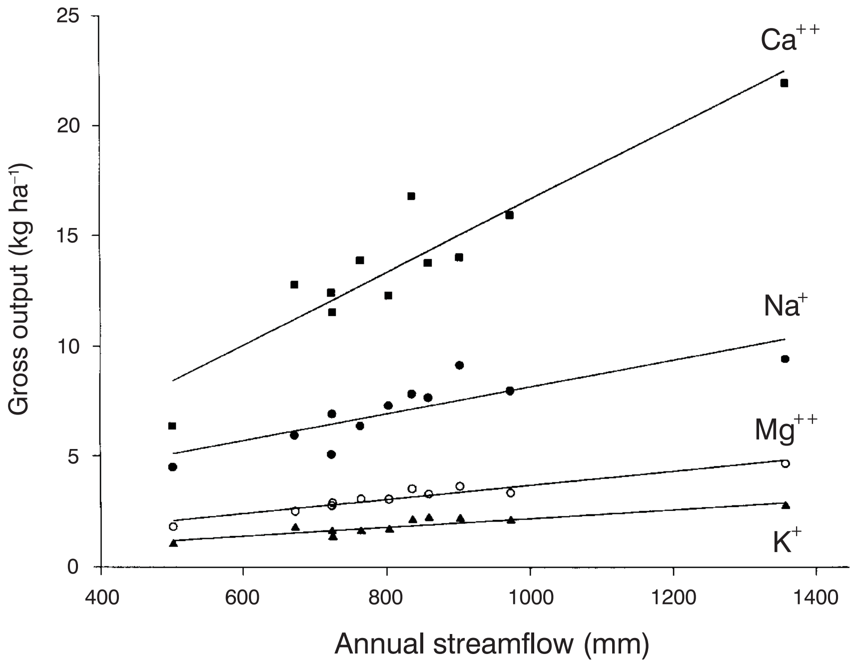

Fig. 15.11. Relationship between annual streamflow and gross output of calcium, sodium, magnesium and potassium (kg ha^{-1}) during 1963–1974 for undisturbed catchments at Hubbard Brook Experimental Forest (after Likens and Bormann, 1995).

rainfall, often hundreds of kilometres inland, and occurring over very long periods of time. Prior to European settlement, substantial parts of Western Australia were covered with eucalypt forest, jarrah (*E. marginata*) being a

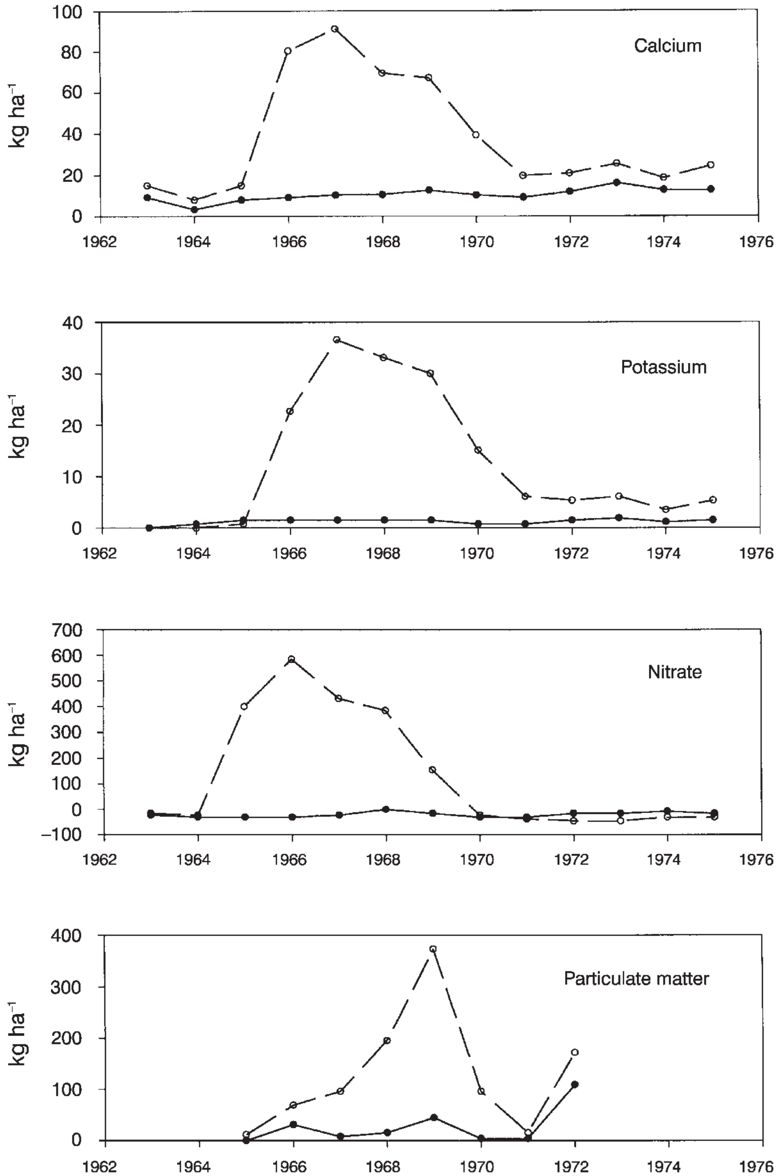

Fig. 15.12. Export patterns of dissolved substances (calcium, potassium and nitrate) and particulate matter in water from a cleared catchment (----) and a control catchment (——) (after Likens *et al.*, 1978).

common species. Considerable forest clearance took place, particularly for growing cereals, in the decades following colonization. The annual water use of the perennial native woodlands and forest is high and recharge rates were very small, rarely exceeding 1–2 mm year^{-1}, but after forest clearing and replacement, typically with annual crops, recharge rates increased 10–100-fold, depending on rainfall and location (Farrington and Salama, 1996). The increased recharge has caused aquifers to fill and water tables to rise, sometimes by as much as 2 m year^{-1}. The groundwater is naturally saline and the rise mobilizes salts in the soil profile. After some decades, saline groundwater, with salt concentrations in excess of that tolerable for plant growth, emerges as saline seeps at the lowest points in the catchments and also discharges to streams (Peck and Williamson, 1987; Schofield and Ruprecht, 1989).

Measures to remedy the salinity problem of low-lying areas and streams have a common aim which is to reduce groundwater levels. One of the options has involved planting tree or shrub vegetation which is likely to emulate, to a semi-quantitative degree at least, the water use of the original vegetation (George, 1990; Farrington and Salama, 1996). Understanding groundwater processes and application of modelling to these Western Australian catchments have indicated options for location of new forests to best reduce the recharge to groundwater. Using groundwater modelling, Salama (1993) showed that reforestation of the upper 25% of catchments would be sufficient to reverse salinity trends. Tree planting will take some years to exert an effect on groundwater recharge. Other short-term, engineering, options can augment the impact of tree planting and will be effective immediately. Measures such as pumping groundwater into evaporation lagoons or establishing shallow soil scrapes (contour banks) which can capture and redirect surface water away from recharge sites are feasible choices.

3.4 Acidification

The sensitivity of a catchment to acid rain is a function of the calcium content of its soils and weathering bedrock; the higher the calcium content, the greater is the capacity to neutralize effects of acid pollutants (acid rain). For the same soil conditions, streamflow is more acid and has higher concentrations of aluminium under forest than under any other vegetation. The differences in water chemistry are most marked at high flows, and typically one unit difference of pH and up to a fourfold difference in aluminium can be observed. There are also other effects observed in acidified catchments which relate to the greater variation in pH and also greater concentrations of chloride, sulphate and nitrate. The increase in the acidity of runoff is related to the greater deposition of pollutants on to the forest canopies because of, usually, greater leaf areas and lower aerodynamic resistances. Although deposition rates are greater to forest than other vegetation, there will be site differences between forests which are related to altitude and rainfall. Simple preventative or remedial measures for stream acidification are elusive. One possibility is to create a treeless buffer zone adjacent to the stream. However, such a measure must be balanced against the positive benefits of trees adjacent to the stream (see below).

3.5 Streamside buffer strips

It is clear that a zone of vegetation adjacent to streams can confer considerable benefits to water quality of the stream and benefit the stream biota (Brooks *et al.*, 1997). The vegetation adjacent to streams can assume a number of forms, while still performing a broadly similar function. One of the simplest forms will be a zone of the commercial forest at the stream edge which remains unlogged but equally the buffer zone could be other streamside vegetation (e.g. fens or bogs). Buffer or filter strips should have high infiltration rates, which coupled with roughness of the soil surface, transforms overland flow into subsurface flow. Sediment from surface erosion becomes trapped, and water borne solutes enter the soil, where uptake by buffer vegetation can subsequently occur.

An important requisite of a buffer strip is that it is sufficiently wide enough to fulfil its protective purpose. The buffer strip should have a minimum width, depending on the slope between the zone of forest disturbance and the stream. With a slope of between 0 and10%, the buffer strip should be at least 15 m and up to 45 m for a slope of between 40 and 70% (Brooks *et al.*, 1997).

Trees along stream banks serve a useful role in mechanically binding the soil, conferring strength to the stream bank, thereby reducing the frequency of slumps and further erosion into the stream. Tree leaves form a major component of the coarse particulate organic matter (CPOM) and an important heterotrophic energy source to stream dwellers. Another component of CPOM, the coarse woody debris (CWD) also provides a source of organic matter but the decomposition of CWD and provision of carbon in streams will be considerably slower than for foliage. However, the CWD serves many other valuable roles. Gurnell *et al.* (1995) emphasized the importance of CWD for controlling sediment and organic matter transport, for channel stability, for physical habitat diversity as well as providing food for a range of biota (from microbes to fish), and shelter, particularly, for fish. Gurnell *et al.* (1995) advocate the active management of streamside trees to ensure the provision of the benefits to the stream and its biota. Gurnell *et al.* (1995) propose that management should emulate natural forest, with a mixture of species, perhaps conifers and broadleaves, providing a variety of food sources and large wood to the stream. Large wood should be left in the stream and light openings should be created by selective thinning as the canopy closes over the stream.

The presence of forests can have a marked influence on the temperature regime of streams. The effects of forest removal are usually greatest on small headwaters and tributaries that may have been completely shaded. In the case of bigger streams there is a greater heat capacity and in natural circumstances they would not be fully shaded. Large rivers, lakes and reservoirs are unlikely to be affected by land use changes at their edges (Satterlund and Adams, 1992). The impact of changing vegetation cover adjacent to small streams will be affected by many factors including, time of year, degree of exposure, streamflow and input of groundwater. Beschta and Wetherred (1984) have developed an algorithm to estimate stream temperatures taking into account the relevant physical variables.

3.6 Stream biota

The implementation of better methods for the management of streamside and in-stream habitat requires effective means to document adverse effects. The relative ease of measurement and standardization of physical and chemical variables has led to the use of biological oxygen demand and other chemical indicators under the assumption that they are useful surrogates (Karr, 1991). A major problem with chemical analysis of stream waters is that it only provides a point description with little information on the dynamics of stream quality or the effects on stream biota. The in-stream fauna and flora reflects the quality of habitat and water which in turn reflects the effects of off-stream management activities. Aquatic biodiversity is therefore a good measure of the success of forest protective management formulas. It would be impractical to monitor and report on the complete ensemble of biological diversity within streams and water bodies; therefore, as a surrogate, a representative subset of the biota should be recorded. The aquatic macro-invertebrates are probably the most useful because they are a biodiverse group comprising a very large percentage of the animal diversity in water bodies, are adequately understood taxonomically and there is a history of successful use for monitoring aquatic ecosystems.

There are a number of monitoring protocols from which the predicted composition of an undisturbed fauna at any monitoring site can be estimated. An index of deviation of the monitored site after comparison with the predicted undisturbed fauna can be calculated. While numerous studies have examined the impact of a wide variety of human impacts such as dam construction and operation, river abstraction and effluent disposal on aquatic macro-invertebrates, fewer studies have been made on the impacts of forest management activities. A research requirement is to test the suitability of national models at regional and smaller scales when the national models have insufficient resolution. Such models would incorporate sampling data from reference sites, especially small streams. The model would be tested for sensitivity to natural and human impacts.

Currently there is a growing interest in a number of biological measures, including invertebrates (Plafkin *et al.*, 1989) and fish-based (Karr, 1981) indices. The Index of Biotic Integrity (IBI) proposed by Karr (1981) is a composite of 10–12 individual measures including species richness and composition, local indicator species, trophic composition, fish abundance and fish condition. Application of IBI requires a good deal of local calibration. Measures of species richness must be based on expected values for streams of a given size and zoogeographic region, and require suitable undisturbed locations to serve as reference sites. The IBI may use fewer metrics and cold water streams may lack some of the species groups. Nevertheless, IBI is a valuable tool because of its capacity to convert relative abundance data of an assemblage of species into a single value of biotic integrity which has been shown to vary with environmental degradation. It would not be practical to monitor and report on the complete ensemble of biological diversity within streams and water bodies. Therefore, as a surrogate, a representative subset of the biota should be monitored. The use of biological monitoring has been advocated with macro-invertebrates or fish commonly proposed as the foci of biological monitoring (e.g. Resh *et al.*, 1996). Table 15.5 summarizes the benefits and disadvantages of using macro-invertebrates to monitor stream quality. Some of these disadvantages will be particularly important in more remote areas where the baseline diversity and taxonomy of macro-invertebrates may still remain to be evaluated.

For biological assessment, the River Invertebrate Prediction and Classification System (RIVPACS) measures water quality on the basis of the known tolerance levels of macro-invertebrate taxa to organic pollution – quality is obtained by comparing the invertebrates present at a site with those expected if the site was unpolluted (Wright *et al.*, 1998). The Mean Trophic Ranking (MTR) system provides a surrogate of water quality with respect to nutrient levels. The MTR system uses aquatic macrophytes to give a trophic score based on the plants present at the site (Holmes *et al.*, 1998)

Another approach to linking stream habitat requirements to species of fish or macroinvertebrates, for example, is to define the amount of habitat available in relation to estimates of the stream discharge. Water flow can be evaluated in ecological terms by the Instream Flow Incremental Methodology which is used in association with the Physical Habitat Simulation (PHABSIM) computer model (Thomas and Bovee, 1993). PHABSIM requires a simulation of river hydraulics, based on field surveys of channel geometry at several transect points, along with measurements of water surface levels and stream velocities at these points under several, differing flow regimes. This hydraulic information is combined with Habitat Suitability Indices for velocity, depth and substrate/cover for chosen target species such as salmonid fish or macro-invertebrates to produce the Weighted Usable Area of the stream or river at a range of flows. It is then possible to produce habitat duration curves based on a series of historical or predicted flows and a probability of a certain type of habitat being available or

Table 15.5. Advantages and difficulties to consider in using benthic macro-invertebrates for biological monitoring (after Resh *et al.*, 1996).

Advantages	
1.	Being ubiquitous, they are affected by perturbations in all types of waters and habitats
2.	Large numbers of species offer a spectrum of responses to perturbations
3.	The sedentary nature of many species allows spatial analysis of disturbance effects
4.	Their long life cycles allow effects of regular or intermittent perturbations, variable concentrations, etc., to be examined temporally
5.	Qualitative sampling and analysis are well developed, and can be done using simple, inexpensive equipment
6.	Taxonomy of many groups is well known and identification keys are available
7.	Many methods of data analysis have been developed for macro-invertebrate communities
8.	Responses of many common species to different types of pollution have been established
9.	Macro-invertebrates are well suited to different types of pollution perturbation
10.	Biochemical and physiological measures of response of individual organisms to perturbations are being developed
Difficulties to consider	
1.	Quantitative sampling requires large numbers of samples which can be costly
2.	Factors other than water quality can affect distribution and abundance of organisms
3.	Seasonal variation may complicate interpretations or comparisons
4.	Propensity of some macro-invertebrates to drift may offset the advantages gained by the sedentary nature of many species
5.	Perhaps too many methods for analysis available
6.	Certain groups are not well known taxonomically
7.	Benthic macroinvertebrates are not sensitive to some perturbations, such as human pathogens and trace amounts of some pollutants

not during a certain length of time. The River Habitat Survey (RHS) method (Raven *et al.*, 1998b) provides a description of 500 m lengths of river based on physical characteristics and degree of habitat quality; stream bank and locality features are also included.

Raven *et al.* (1998a) discuss recent moves towards a more integrated approach to river basin management because of international concensus that environmental quality, as a whole should be approached in a comprehensive way. They point out that there are a number of common attributes recorded by the methods such as RIVPACS, PHABSIM, MTR and RHS and call for attempts to harmonize the various river classification and evaluation methods.

4 Conclusions

4.1 Streamflow quantity and timing

- The impact of changes in forest cover, deforestation or afforestation is evaluated traditionally by catchment studies over a few to many years. Changes in catchment response to deforestation may take a few years to stabilize because of responses due to changes in the soil surface by logging or because residual soil water deficits following the forest cover need to be satisfied.
- After initial equilibration, the catchment response to deforestation or afforestation will need to be evaluated against the background of natural variation in rainfall.
- In the large majority of cases removal of forests from catchments leads to an increase in streamflow and afforestation causes decreases in streamflow. Exceptions are tropical montane cloud forest and possibly deciduous forests in low rainfall areas.
- The greatest impact on streamflow – increases or decreases – occurs with removal or planting of areas of evergreen, fast-growing trees. Often this will mean pines and eucalypts. The largest changes being up to 70 mm for a 10% change in forest cover but values around 40 mm for a 10% change in forest area are more typical for evergreen fast-growing forest cover. Comparable values for deciduous broadleaf woodlands would be around 20 mm for a 10% change in forest cover.
- The impacts on streamflow of changes in forest cover of less than 20% may be difficult to detect statistically by catchment studies.
- A range of sophisticated distributed process models are now available to predict catchment behaviour to changes in forest cover. Vegetation parameters may need to be acquired for unfamiliar vegetation. The adequate spatial representation of rainfall and soil hydraulic conductivity for catchments is a problem for distributed process models.
- A number of studies in which streamflow from forested and non-forested catchments has been compared have shown that the seasonal patterns of streamflow do not deviate proportionally from differences observed in annual streamflow totals. There are examples, however, in which the pattern of streamflow timing has differed between the two land covers. Unfortunately, the causes of the change in timing of streamflow, mostly, remain obscure. A difference may reflect the low water use by the short vegetation during the dry dormant period, at a time when forest water use is maintained.
- Changes in streamflow can be related to establishment of forest cover, maximum growth rates and declines in growth rate, as forests age. Surrogates for high growth rates, net primary production and canopy capacity such as stand basal area or stand basal sapwood area offer a

useful indicator of relative differences in forest water use as forest stands age or even between species.

4.2 Water quality

- On an annual basis export of individual inorganic ions from undisturbed forests increases linearly with annual precipitation. These relationships are very robust for individual forests but differences exist between forests. Nutrient exports are greater from forests growing on nutrient-rich soils. The major nutrient, in terms of mass exported, from both undisturbed and disturbed forests is nitrogen.
- Forest harvesting leads to a large increase in export of nutrients to streams. A number of factors influence the loss of nutrients: release of inorganic material from harvesting residues, leaching from disturbed soil and an increase, because of reduced forest evaporation, of soil water available for drainage to the streams. Burning of forest residue after harvesting will make inorganic material more available for leaching and will enhance runoff because of reduced infiltration.
- The increase in nutrient concentrations in streams following forest harvesting returns to pre-harvest levels after 3–4 years. Regrowth vegetation at logged sites; ground flora, resprouts, or young trees will be strong sinks for available nutrients.
- The major contribution to contamination of forest streams following logging comes from suspended sediments. Increased turbidity caused by sediments in suspension seriously degrades the light levels in streams for fauna and flora and disrupts feeding and respiration. Fine sediment deposited on the forest floor can impoverish spawning sites for fish.
- Sediment production during logging activities is enhanced by rainsplash erosion on areas from which the protective cover of trees, understorey and litter has been removed or reduced. These areas are harvesting zones, log landings, skidding trails and roads. Compaction of these areas also promotes runoff and associated sediment movement. Unpaved roads, particularly, are a major source of sediment.
- A number of proposed measures to prevent sediment movement from tracks and roads into streams during harvesting are reviewed. An adequate buffer zone between the stream and the end of the skidding trails should be provided. Frequent cross drains for logging roads should be installed.
- After harvesting has been completed, cross banks or earth bars across tracks should be at frequent intervals to prevent large separate discharges amalgamating in adjacent receiving areas. Closure and breaking up of the surface of abandoned logging roads should be considered.
- Buffer zones adjacent to streams are often proposed as a panacea for a number of problems related to forestry and tree harvesting. Some of

the roles for buffer zones are: preventing sediment, nutrient and acidic deposition from entering streams, provision of coarse woody debris and by being a site for trees which will moderate temperature regimes at the stream edge and in the stream. Some of these roles might be conflicting and there is uncertainty about the optimum widths for buffer zones.

5 Indicators

All the indicators below relate to the criterion 'Conservation and maintenance of water resources'. This criterion is Number 4 in the Montreal Process (Grayson and Maynard, 1997). Indicators are separated firstly into those mainly related to the quantity of water and secondly, the quality of water.

5.1 Quantity and timing of streamflow

Two coarse indicators of the sustainability of catchment streamflow are:

- *the percentage cover of forest in the catchment area*;
- *the fraction of the forest cover in the catchment area which is conifers, broadleaf evergreen or deciduous forest*.

These indicators gauge in an approximate way the degree and nature of forest cover acknowledging that, in the large majority of cases, forests will use more water than short vegetation. The second indicator relates to general trends in water use, conifers > broadleaves. The indicators serve to emphasize options available to improve streamflow.

Finer scale indicators could be:

- *basal area (or sapwood basal area) of the forest stand as a proportion of the basal area (or sapwood basal area) of the stand at its most productive age*. In established stands the age of the forest has been identified as a major source of variation in water use. This indicator aims to show the productivity of the stand relative to the productivity at the optimum age. The information would allow an assessment of how close to peak water use the stand is. Basal area is chosen because of the ease with which it can be measured under field conditions. There are other equally useful measures of stand productivity, e.g. sapwood basal area or leaf area index. Typically these are somewhat less easy to monitor in field conditions. Also, as forests are established on catchments previously without forest the increases in basal area relative to the maximum possible for that species in those conditions will be a sign of how rapidly the forest is approaching the maximum productivity which is being equated with maximum water use.
- *fraction of land, which has forest cover, which is on low-lying land at stream margins*. The presumption with this indicator is that forest growing on

the wettest land of the catchment will have the greatest opportunity to influence recharge to the stream and will have optimum water use because water supplies will be unlimited. There are two considerations to be made about this indicator. In this chapter there has been discussion about tree species with conservative water use because of negative feedbacks of leaf stomatal conductance with atmospheric humidity. It is likely that these trees could not exploit the abundance of water at the stream side. A second point is that an indicator below is proposed because it is envisaged that stream side buffer zones have considerable ecological benefits, and in some cases the presence of trees in the buffer zone conveys further benefits; providing coarse woody debris to the stream and preventing excessive stream temperatures by providing shade.

- *fraction of land which is afforested (or deforested) which was previously populated (or is likely to be populated) by dormant, short vegetation in the dry season*. The understanding of the controls of deviations in low and peak flows in response to changes in forest cover is less developed than for annual streamflow differences caused by afforestation of forest clearing. When forest replaces, or is replaced by, short vegetation, which dies in the dry season or becomes dormant, the seasonal water use of the short vegetation will be reduced proportionally more than that of the forest. This indicator, which at this stage should be regarded as interim, reflects the seasonality of water use by short vegetation on catchments.

5.2 Water quality

Two coarse indicators of the sustainability of catchment stream water quality are:

- *the percentage of forest area occupied by roads, skid trails and log landings*. This indicator, in addition to others aims to reflect the relative rates of sediment delivery to streams and therefore relative stream water quality. Attempts to quantify the impacts of roads simply by calculating a density of road length per forest area are problematic, because intrinsically road density can be very low and therefore sensitive to small absolute differences in road length.
- *the proportion of forest stream length that is protected by riparian buffer strips of adequate width*. This provision of buffer zones serves a number of purposes in maintaining stream quality and their ecological systems. Included in these benefits is the prevention of sediment from logging activities and roads entering streams, but also buffer strips can include stretches of light tree cover to moderate stream temperatures. Buffer strips can also provide a source of coarse woody debris for the stream.

Finer scale indicators could be:

- *the density of stream crossings and contiguities (streams and roads adjoining without adequate buffer strips) within the forest area (number per km^2)*. It is not necessarily the road length or density *per se* that is the critical factor but the proximity of roads to streams and the frequency of road crossings of streams and this indicator is directed at this point.

The overall aim for sustainably managing stream and river systems is to maintain the stream ecological systems rather than to maintain quality itself. Therefore an indicator is proposed which gauges the diversity of stream biota or the health of the stream.

- *locally/regionally derived models based on predicted macro-invertebrate occurrence*. Macroinvertebrates have been selected to represent maintenance of biotic diversity of the stream. Other animal or plant groups could have been selected or a combination of groups allied with physical attributes of the stream could form the basis of the indicator.

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The Role of Forests in the Global Carbon Cycle

16

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Forests play an important role in the global carbon cycle, with estimated carbon stock in forests (990 GtC) being of comparable magnitude to that contained in the atmosphere (750 GtC). Of the carbon contained in forests, about two-thirds reside in soil organic matter and only one-third in vegetation. In the tropics, 13 million ha of forests are converted to other land uses every year, leading to an estimated carbon release of 1.6 GtC year^{-1}. At the same time, forest cover is estimated to be increasing at higher latitudes by about 1.2 million ha year^{-1}, thereby sequestering 0.7 GtC year^{-1}. Forest harvesting for fuelwood and wood and paper products is estimated to lead to an annual release of about 1 GtC year^{-1}, with about half of that used for fuelwood. After allowing for losses during production and decay of previously produced wood products, it is estimated that the global pool of carbon in wood and paper products is increasing by about 140 million tC year^{-1}.

Forests can be managed to maximize carbon stocks by preserving existing forest cover, especially where forests have a high standing biomass, and by establishing new forests on currently non-forested land. It is also possible to modify forest management in order to maximize site carbon storage. However, there is a limit to the potential contribution that on-site carbon storage can make to global carbon cycles because the stocks of carbon stored on any area of land are obviously finite. However, forests can make an ongoing and sustainable contribution to reducing net carbon dioxide (CO_2) emissions through the substitution of wood for fossil fuels, either in energy generation, or by substituting wood for other materials that require larger CO_2 emissions in their manufacture. Bio-energy

currently supplies about 14% of the world's primary energy needs, and it has been estimated that bio-energy has as much potential to contribute to the world's energy supply in the near future (by 2025) as all other forms of renewable energy combined.

While these pools and fluxes constitute significant components of the global carbon cycle, there are no easily measurable direct indicators by which to judge whether specific forests are net sources or sinks of carbon. Net carbon flux is essentially determined by the difference between carbon uptake in growth and losses in harvesting, fire and decomposition of dead organic matter. Precise measurements of forest growth are difficult because changes are typically only small fractions of standing biomass, because growth is greatly affected by year-to-year variability in climate and biotic factors, and because many forests are highly heterogeneous. Measurement of changes in soil organic matter is even more difficult. Even when trends in carbon stocks have been established, it remains uncertain whether they reflect typical conditions, and whether these trends will persist with variation in climatic and disturbance regimes. Changes in the carbon stocks of any region may be assessed by a combination of methods, including growth modelling, remote sensing, ground-based measurements including harvesting statistics and atmospheric trace gas studies.

1 Introduction

Forests are one the world's major carbon stores, containing about 80% of all above-ground terrestrial biospheric carbon and 40% of terrestrial below-ground carbon (Kirschbaum *et al.*, 1996). Carbon in above-ground biomass of individual forests can be released during wildfires, by decomposition of dead biomass after mortality which may be caused by normal ageing or drought, pest or disease outbreaks or by random catastrophic events, such as storms. Between such disturbances, most forests grow larger and absorb carbon from the atmosphere. These factors largely balance out so that under natural conditions the large carbon pools in forests change little. However, large-scale anthropogenic disturbance through logging, land-use change (e.g. clearing of native vegetation; Fig. 16.1), and inadvertent wide-spread pollution, has meant that forests are now no longer in equilibrium, and forests in individual regions have become significant sources or sinks of carbon.

As a result of logging, land-use change or changed growth in response to changes in environmental conditions, carbon stocks in forests can change greatly and significantly affect atmospheric CO_2 concentrations (Schimel *et al.*, 1996). Hence, the 'maintenance of forest contribution to global carbon cycles' has been recognized as one of the criteria for sustainable forestry under the Montreal Process.

Through exercising different management options, forests can be managed to maximize carbon storage in the biosphere (Brown *et al.*, 1996). This can involve both preservation of existing forests with high carbon stocks per unit area, and afforestation or reforestation of currently unforested land. It can involve both the establishment of environmental plantings in perpetuity and commercial plantations over short rotations. Potentially the most sustainable contribution can be made by forests if wood is substituted for fossil fuels, either by substituting wood for fossil fuels as an energy source or by replacing alternative materials that lead to larger CO_2 emissions in their manufacture.

Fig. 16.1. A semi-arid eucalypt woodland in central Queensland that is used predominantly for cattle grazing. Such forests have been cleared to enhance grass growth and stock-carrying capacity, or for conversion to cropping. Such extensive land-use change can have negative impacts on biodiversity and greenhouse gas balance, and can induce dryland salinity over extensive areas. Indicators are needed to quantify these impacts, and to guide policy and management decisions.

The respective contribution of forests to global carbon cycles is different under each of these options, and different issues are of key importance in each case. In the following, the significance of these options will be put into perspective and some of the key issues and interactions will be discussed.

2 Existing carbon stocks in forests

Carbon is exchanged between several pools in the global carbon cycle (Fig. 16.2). The exchange with deep oceans is critically important for the long-term equilibrium of atmospheric concentrations, but the rate of exchange is very slow. Exchange between the atmosphere and living vegetation is faster, and there are also important fluxes between forest vegetation, wood products and the atmosphere (Fig. 16.2).

There are still vast reserves of fossil fuels in the Earth's crust although most are not recoverable with current technology. If means can be found to efficiently extract them, however, their cumulative emissions will increase the atmospheric concentration of CO_2 many times. Eventually, the deep oceans will become a permanent storage pool for most of that carbon, but with the rate of exchange between the atmosphere and the deep oceans being very slow, temporary storage in terrestrial biomass attains an important role. Wood products (fuelwood and commodities) are currently being removed from forests at an estimated rate of about 1.1 GtC year^{-1}, with the pool of woody products increasing by about 140 MtC year^{-1} (Winjum *et al.*, 1998; see

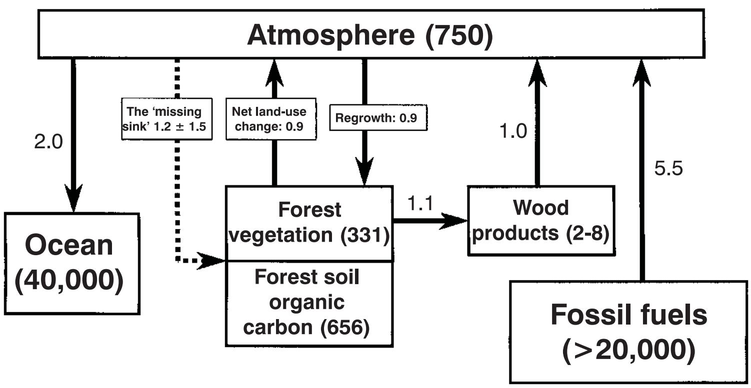

Fig. 16.2. Estimated sizes of pools and fluxes in the global carbon cycle that are relevant to forestry. Pool sizes are given in GtC. Numbers next to arrows give estimated net fluxes in GtC year^{-1}. Tables 16.1 and 16.3 and Fig. 16.4 give further details and sources of information. Arrows drawn in solid type are based on direct estimates, whereas the 'missing sink' is calculated from the difference remaining after other pools and fluxes have been explicitly quantified.

below). After logging, most forests are replanted or allowed to naturally regenerate so that forest cover will eventually be re-established. However, the replacement of many old-growth forests containing large amounts of carbon by younger forests containing less carbon is resulting in an ongoing reduction in forest biomass in managed forests (Houghton, 1998).

Forests currently contain an estimated 987 GtC (Fig. 16.2; Table 16.1) of which two-thirds is contained in soil organic matter (SOM; Brown *et al*., 1996). The proportion of total site carbon that is contained in SOM increases with latitude. In tropical forests, there are about equal proportions of carbon in vegetation and SOM, whereas some boreal forests contain more than 10 times as much carbon in SOM as in vegetation (Brown *et al*., 1996).

Table 16.1. Estimated sizes of some relevant global carbon pools and fluxes (in 1990).

Atmosphere[a]	750 GtC
Atmospheric increase[a]	3.3 GtC year^{-1}
Ocean[a]	40,000 GtC
Fossil fuels (conventional)[a]	3,500 GtC
Fossil fuels (unconventional)[b]	> 20,000 GtC
Global forest carbon stocks (1990)[c]	987 GtC
Plant material (incl. roots)[c]	331 GtC
Soil organic matter[c]	656 GtC
Net change in forest stocks 1980–1990[c,d]	–0.9 GtC year^{-1}
Net change in high-latitude forests (> 50°)[c,d]	+0.48 GtC year^{-1}
Net change in mid-latitude forests (25–50°)[c,d]	+0.26 GtC year^{-1}
Net change in low-latitude forests (< 25°)[c]	–1.65 GtC year^{-1}
Cumulative net carbon emission due to land-use change (1800–1990)[e]	–124 GtC
Carbon emission rate from fossil fuel combustion (1980–1989)[a]	–5.5 GtC year^{-1}
Cumulative carbon emission from fossil fuel combustion up to (1800–1990)[f]	–212 GtC

[a]Schimel *et al*. (1996).
[b]Nakicenovic *et al*. (1996). 'Conventional' includes coal, oil and gas recoverable by conventional means, including known reserves and additional resources estimated to be discovered with 50% probability. 'Unconventional' refers to estimated large additional resources that are not recoverable by current conventional means.
[c]Brown *et al*. (1996).
[d]Brown *et al*. (1996) calculated mid- and high-latitude forests to be increasing at 0.74 GtC year^{-1}, whereas Schimel *et al*. (1996) attributed only 0.5 GtC year^{-1} to that source.
[e]Houghton (1999).
[f]Marland *et al*. (1999).

Overall, forests are estimated to be losing about 0.9 GtC year^{-1} to the atmosphere (Fig. 16.2; Table 16.1), consisting of losses in the tropics and gains at higher latitudes. Forests in the tropics are estimated to lose about 1.65 GtC year^{-1} due to ongoing land clearing. The Food and Agriculture Organization of the United Nations (FAO) reported that natural forests in the tropics were lost at a rate of 14.6 million ha year^{-1} between 1980 and 1990 and at 12.9 million ha year^{-1} between 1990 and 1995 (FAO, 1997). This constituted an annual loss of about 0.7% year^{-1} of the remaining natural forest area.

At the same time, there are believed to be gains in temperate and boreal regions due to reforestation and increase in the average size of existing forests (Armentano and Ralston, 1980; Kauppi *et al.*, 1992; Dixon *et al.*, 1994; Brown *et al.*, 1996). Forest area is reported to have increased by 1.2 million ha year^{-1} between 1990 and 1995 (FAO, 1997). Forest size is increasing because many forests have been replanted after heavy logging earlier during this century and are now still in their early growth phase.

Overall, conversion of forests to agricultural land has added to a cumulative loss of about 124 GtC between 1800 and 1990 (Houghton, 1999). The early loss came mainly from the temperate region, with the tropics having been the major source since about 1945 (Houghton, 1995). It is likely that there are additional unspecified sinks in the biosphere (the 'missing sink') required to close the global carbon budget (Schimel *et al.*, 1996). These sinks could reside in increasing forest biomass in regions with poor inventory statistics, or they could correspond to increases in SOM brought about by increasing CO_2 concentration or nitrogen fertilization through industrial pollution (e.g. Kirschbaum, 1993). Temperature increases over this century, however, are more likely to have caused losses of soil organic carbon, especially in naturally colder regions (Kirschbaum, 1993, 2000; Wang and Polglase, 1995). Studies of the spatial distribution of CO_2 concentrations across the globe also point to the tropics as a likely site of additional unidentified carbon sinks (e.g. Enting and Mansbridge, 1991).

For comparison, fossil fuel carbon was emitted in the 1980s at a rate of 5.5 GtC year^{-1}, for cumulative emissions of 212 GtC between 1850 and 1990. Hence, the majority of atmospheric CO_2 increase comes from fossil fuel burning, both at present and historically. None the less, changes in the world's forests can make a significant additional contribution to either increase or decrease the net flux of CO_2 into the atmosphere by adding or reducing carbon stocks in vegetation, or by substituting for fossil fuel use.

3 Preserving existing carbon stocks

In aiming to maximize the beneficial contribution of forests to the global carbon cycle, the critical aspect is the total amount of carbon held in forests, not the rate at which carbon is taken up by forests. Any amount of carbon that is contained in vegetation is an amount of carbon that might otherwise be in the atmosphere where it would contribute to global warming. Mature forests generally contain a larger amount of carbon in their stems than young forests even though the young forests may be rapidly growing and absorbing carbon at a faster rate than old forests. If the young forests continue to grow, they may, of course, become important and useful storage pools over time.

Once forest harvesting commences in a new area, it generally leads to a release of carbon to the atmosphere as forests that have attained a certain average standing biomass under natural conditions are drawn to a lower average standing biomass through repeated wood removal. This is illustrated in Fig. 16.3.

This illustrative simulation shows the effect of forest harvesting on the carbon stocks contained in the biomass of a hypothetical forest estate. The change in carbon stocks is then used to compute an annual rate of carbon loss. These simulations assume that forest operations commence at year 1, that 1% of the total area is logged each year and that each coupe is regenerated with the same forest type as before. Forest growth is calculated with a standard forest growth equation with growth reaching a maximum after a few years and then decreasing progressively, with little further growth for stands 100 years old. All other disturbances, such as fire or wind throw, are assumed to continue in the managed forest in the same way as in the previously unmanaged forest.

As forestry introduces an additional disturbance into the forest system, the forest's total carbon stocks decline. Each year, a small part of the mature forest area is harvested and replaced by regrowth forest. Even though the total forest area remains unchanged, its average carbon stocks are reduced. If there were no further wood removal, overall carbon stocks would eventually return to their previous value. If, however, a different coupe is harvested each year, overall carbon stocks of the forest estate as a whole continue to be reduced.

Carbon loss is greatest in year 1 because in that year, there is only a carbon loss due to harvesting. In year 2, the overall net loss is slightly reduced because the loss due to the coupe harvested in year 2 is partially compensated by new growth in the coupe that had been harvested in year 1. In subsequent years, more and more regrowing coupes increasingly compensate for ongoing carbon losses due to harvesting.

If the cutting cycle is repeated after each rotation, a new equilibrium is eventually established. In the example given, the new steady state is reached after 100 years, but the forest contains only about 50% of the carbon stocks of the original forest. Once the new equilibrium is reached at year 101, there is no longer any loss of carbon as ongoing wood removal is balanced by the same amount of regrowth. This corresponds to the change from harvesting mature forest coupes with high carbon stocks, as was done for the first 100 years, to harvesting regrowth forests from year 101 onwards. In this example, forests are assumed to have reached 83% of their potential carbon stocks at age 100.

The notional pattern in Fig. 16.3 is also evident in global carbon release from forest operations (Fig. 16.4). Houghton (1998) compiled data for the annual rate of net carbon flux due to land-use change and forest harvesting. The calculated numbers are based on carbon loss in logging minus the subsequent carbon gain in forest regrowth. Logging activity is reported to have increased about fivefold over the past 140 years. Despite this large increase in wood removal, the rate of carbon loss was calculated to have been almost constant over that period as more and more regrowing forests compensated for

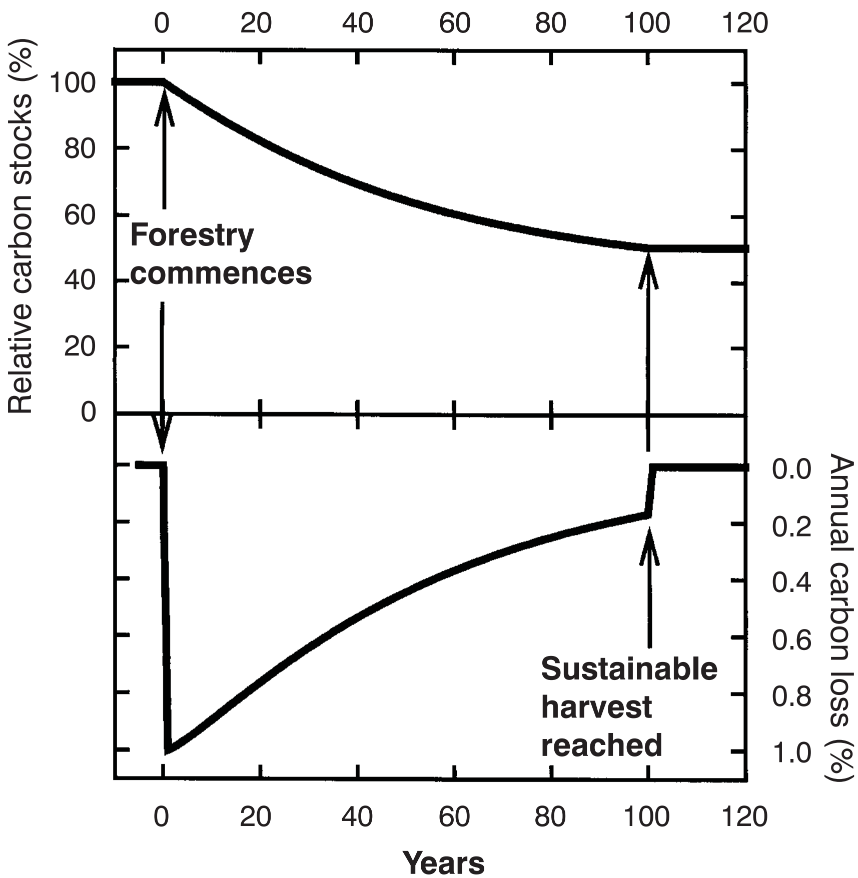

Fig. 16.3. Effects of harvesting on carbon stocks and annual rates of carbon loss. Carbon stocks and loss rates are expressed on a relative scale, with '100%' representing the carbon contained in the whole forest estate upon commencement of forest activities. Carbon stocks are calculated with the Richards equation as:

$$C = [1 - \exp(-0.02\ t)]^{1.25}$$

where C is total site carbon content and t is time in years. The parameters '0.02' and '1.25' are empirical.

the increasing rate of carbon loss in logging (Houghton, 1998). If the rate of wood removal had been constant then the carbon flux associated with logging would eventually tend to zero (as in Fig. 16.3), but while the rate of wood harvesting continued to increase, it caused an ongoing reduction in the standing biomass in forests and a net flux of carbon from forests to the atmosphere.

At the same time, the rate of carbon loss from forest harvesting is much smaller than the rate of carbon loss from land-use change (Fig. 16.4). Upon land-use change, a forest with high biomass is replaced with a different vegetation type that contains much less biomass, leading to a large carbon loss per unit area. In normal forest operations, on the other hand, forests are allowed to regrow so that eventual net changes are small.

As shown above, harvesting old-growth forests must almost always reduce carbon stocks in forests. Consideration of the carbon contained in wood products lessens that conclusion to some extent, but does not nullify it because only a small fraction of wood tends to be stored in products with long lifetimes (see below). However, forestry could reduce the draw-down of carbon stocks in forests if other disturbance agents, such as intense fires or insect outbreaks, could be reduced through forest management, or if existing forests were replaced with different species with higher carbon storage potential.

Some existing old-growth forests have also been severely degraded by past fires, pests or diseases and may contain little carbon in their biomass. Harvesting such forests may then not lead to the same loss of carbon, or could even lead to increased carbon storage if replacement vegetation can remain

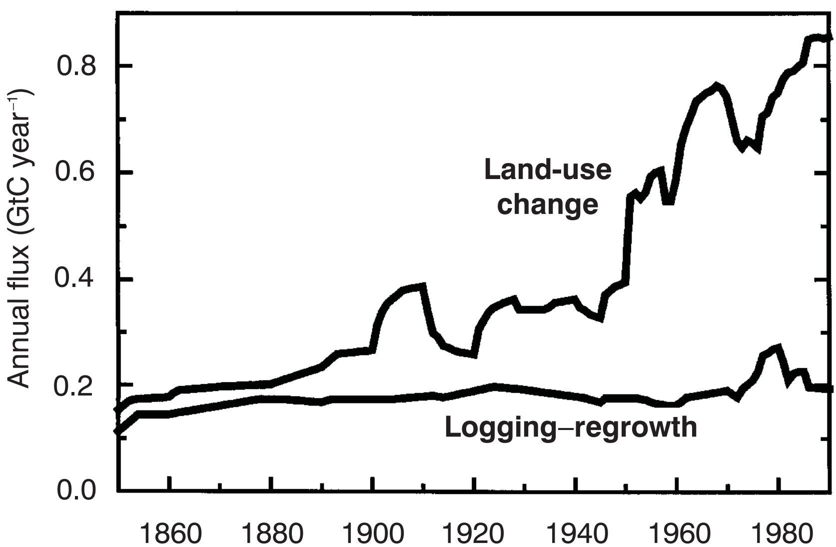

Fig. 16.4. Net carbon flux to the atmosphere from land-use change and logging minus regrowth (redrawn from Houghton, 1998).

free from these disturbance agents. Carbon stocks could then be even higher in a sustainably managed, and repeatedly logged, forest than in the original stands. However, that is likely to be the case only where the original vegetation had been severely degraded.

4 Creating new carbon stores

New forests can also be established on land that was previously non-forested. Such new plantings can be established on land that either was never occupied by trees in the past (afforestation) or on land that had been cleared of trees at some earlier period, usually for agriculture (reforestation). Forests are now often re-established because of better economic returns from tree plantations than from agriculture, to prevent land degradation, to re-create biological habitats, for visual amenity or for local climate amelioration.

The benefit of storing carbon for climate-change mitigation creates an additional incentive to establish new plantings. The most cost-effective plantings for carbon sequestration may often be possible where financial benefits from carbon sequestration can be combined with profits from commercial wood growing (e.g. Kirschbaum, 1996). However, substantial increases in the area under forests will not be possible without creating serious land-use conflicts. Land with best productive capacity is generally the most desirable for the purposes of producing both food and wood. Such land is also generally the most expensive, so that costs of carbon sequestration may be high.

There are, however, opportunities where the different requirements of different activities may create possibilities to sequester carbon cost-effectively. For example, steep hillsides or sites remote from potential markets may be unsuitable for agriculture and commercial forestry, but that would create no obstacle to using those sites for carbon storage. Similarly, agroforestry may be desirable to protect the quality of agricultural land. The establishment of additional trees could then serve several purposes: storing carbon, enhancing agricultural productivity on adjacent land, adding to visual amenity and creating wildlife habitat. The most cost-effective ways to sequester carbon may sometimes be found in conjunction with commercial objectives and sometimes in non-commercial plantings in perpetuity (e.g. Kirschbaum, 1996).

In terms of overall productivity, even the requirements of commercial forestry sometimes coincide and sometimes conflict with the aim of maximizing forest carbon stocks. This is illustrated in Figs 16.5 and 16.6.

Figure 16.5 was drawn using a standard forest growth curve and shows the amount of standing wood over time, the derived current and mean annual increments and the mean amount of standing wood up to respective times. Mean annual increment is the measure most relevant for commercial forestry as it gives the average growth rate from the time of plantation establishment up to possible harvest dates.

Mean standing wood, on the other hand, is the measure most relevant in terms of carbon storage. When a forest is young it makes little contribution to storing carbon, but as it gets older its contribution increases. The forest's total contribution over some period of time is then made up of the less valuable initial low-storage period and the more valuable high-storage period later on. To calculate a forest's total contribution, one can calculate the average amount of carbon contained over all years up to the present. The bottom panel

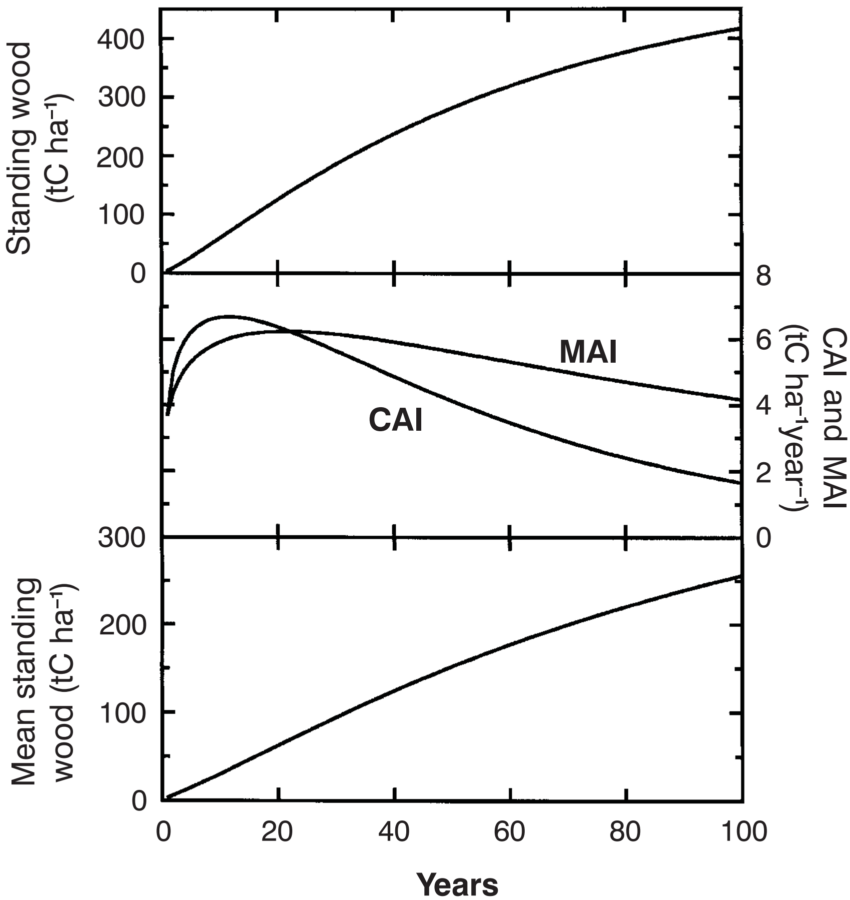

Fig. 16.5. Notional plantation, showing standing wood over time, current (CAI) and mean (MAI) annual increments and mean standing wood for different periods up to a 100-year period. All numbers are expressed in units of tC ha^{-1}, calculated with the Richards equation as:

$C = 500\ [1 - \exp(-0.02\ t)]^{1.25}$

with the same parameters as in Fig. 16.3, but maximum carbon stocks were set to '500' rather than '1' as was done for Fig. 16.3.

in Fig. 16.5 shows that the mean amount of standing wood increases with time, whereas mean annual increment reaches a peak after about 20 years and decreases thereafter.

This is illustrated more directly in Fig. 16.6, in which mean annual increment is plotted against mean amount of carbon contained in wood (from the data in Fig. 16.5). For the first 20 years, both mean carbon content in wood and mean annual increment increase together. Hence, increasing rotation lengths, from say 15 to 20 years, would result in an increased mean amount of carbon stored and increased mean annual increments.

However, mean annual increments reach their maximum before age 25 and decrease thereafter, whereas mean carbon contents continue to increase.

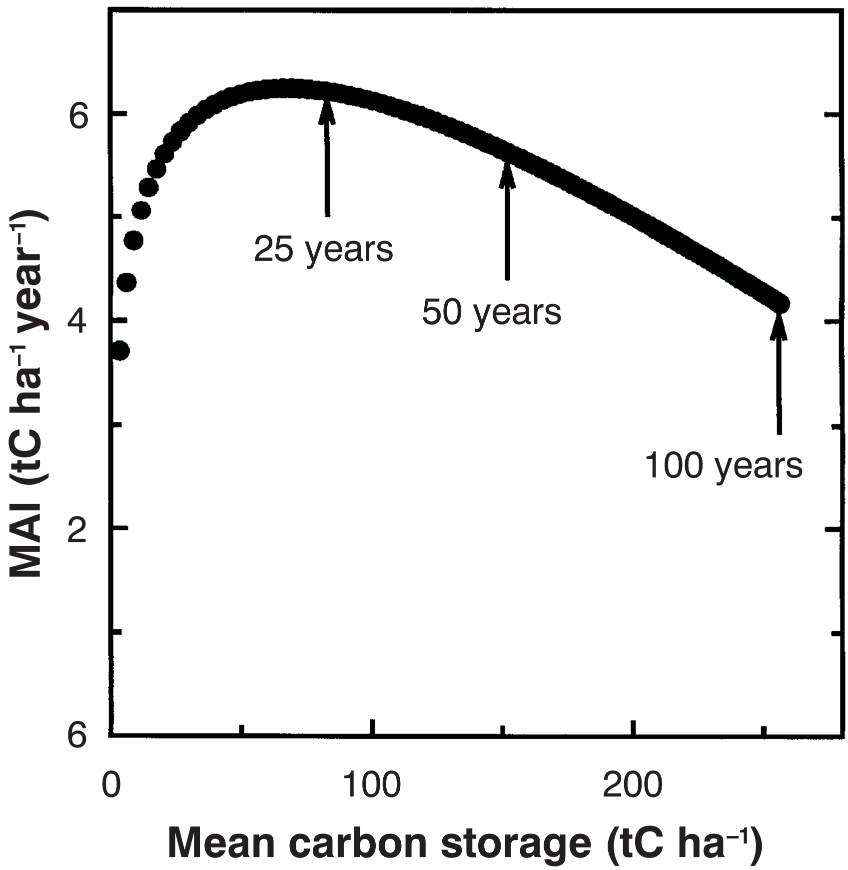

Fig. 16.6. Mean annual increment (MAI) plotted against mean amount of carbon stored in wood over different rotation lengths. Arrows in the figure indicate particular values reached at respective times after planting. This figure was redrawn from the data in Fig. 16.5.

Hence, the requirements for maximizing carbon storage and maximizing growth rates start to diverge, and a compromise must be found between these competing requirements (Kirschbaum, 1996).

5 Soil carbon

The assumption is usually made that increasing above-ground biomass is associated with increased carbon stocks below ground. Hence, based on a number of studies (e.g. Veldkamp, 1994), it is generally believed (e.g. Detwiler and Hall, 1988) that forest vegetation is inevitably associated with greater amounts of soil organic carbon than grasslands. For example, the IPCC Greenhouse gas inventory default assumption is that forest clearing leads to a decrease in SOM.

There appears to be ample evidence that when forests are converted to cultivated croplands, soil carbon amounts indeed decrease (e.g. Detwiler, 1986; Schlesinger, 1986). If the soil is not cultivated, however, soil carbon amounts may not change over decades after transitions between different vegetation types (e.g. Lugo and Brown, 1993; Christopher *et al*., 1997; McGilvray, 1998), and after centuries to millennia, soil carbon amounts often even tend to be lower under forests than grasslands (e.g. Almendinger, 1990). The association between vegetation cover and amounts of SOM is also not very tight, and trends in soil organic carbon may instead depend critically on the condition of either vegetation type. Degraded landscapes often have reduced amounts of SOM. There may then be no consistent change in soil organic carbon following changes between vegetation classes if land degradation can be avoided (e.g. Trumbore *et al*., 1995; Fearnside and Barbosa, 1999).

It is also important whether amounts of soil carbon change under normal forest operations. Johnson (1992), in a broad review of available literature, concluded that normal forest operations had no consistent effect on SOM. The only trends apparent were possible gains in SOM upon fertilization or inclusion of legumes, and losses in response to soil cultivation before replanting. Fire of moderate intensity also appeared to have little effect on SOM (Johnson, 1992), although significant losses have been documented following more intense fires (Raison *et al*., 1985). Because of the large areas of forest that are potentially subject to varying fire regimes, even small changes in the abundance of organic matter in soil could have a potentially large aggregate effect over a whole forest estate.

6 Length of carbon storage in vegetation

When considering the benefits of temporary carbon storage in vegetation, the question of time horizons arises. A plantation that is established for the purpose of growing wood for paper making, for example, stores carbon only for some years before the carbon is released back into the atmosphere. For climate-change mitigation, that plantation is therefore not as useful as a plantation that stores carbon in perpetuity. However, the commercial plantation can still help to lower atmospheric CO_2 concentrations for some period of time.

How can one compare the usefulness of carbon savings by reduced fossil fuel consumption through, say, improved energy efficiency, with temporary carbon storage in vegetation sinks? Similarly, how can one compare the value of short-rotation commercial plantations with environmental plantings in perpetuity? Carbon saved due to improved efficiency or in a planting in perpetuity is a timeless saving, whereas carbon sequestered in a commercial plantation is saved only for the limited time until the plantation is harvested and carbon from wood products is returned to the atmosphere. The cost-effectiveness of these different options can be compared only if they are expressed in the same units.

For example, it may cost \$1000 to improve the energy efficiency of a factory to save 100 tC, or to plant an area of forest to store 100 tC in perpetuity. The cost would be \$10 tC^{-1}. As an alternative, establishing a commercial plantation that can store an average of 100 tC over 50 years may be considered. After allowance is made for profits from the eventual sale of wood, this might require a further subsidy of \$1000. So, the cost would be \$0.2 tC^{-1} $year^{-1}$.

As they are, these respective costs cannot be compared because they are expressed in different units. However, it is possible to compare them if one assigns an arbitrary time horizon over which the 'timeless' saving is to be counted. For temporary carbon storage, the time component is inherent.

If a time horizon of 100 years was used as an accounting horizon, the indefinite planting project would be calculated to have a cost-effectiveness of \$0.1 tC^{-1} $year^{-1}$, which allows a direct comparison with the commercial plantation. The relative cost-effectiveness of different projects is strongly affected by the chosen accounting horizon (Table 16.2). If a shorter arbitrary time horizon were chosen it would increase the attractiveness of short-rotation vegetation sink options that have shorter inherent storage times. On the other hand, if a longer arbitrary time horizon were chosen, storage options in perpetuity would become increasingly more attractive in comparison.

Table 16.2. Correspondence of costs for indefinite carbon savings into different arbitrary accounting periods. These calculations assume a cost of \$10 $(tC)^{-1}$.

Time horizon (years)	Cost \$ $(tC)^{-1}$ $year^{-1}$
50	0.2
100	0.1
250	0.04
1000	0.01

The choice of accounting horizon is essentially arbitrary, but 50–250 years, and most likely 100 years, would seem to be the most 'sensible' as this is within the time period over which problems from global warming are likely to arise. One hundred years is also consistent with the IPCC default time frame for Greenhouse Warming Potentials and still short enough to be within society's planning horizon. Time horizons of less than 50 years would be effectively shorter than the time frame over which climate-change problems are likely to develop, and time horizons longer than 200 years might be considered to be too long for planning purposes.

For vegetation plantings in perpetuity, it would also be difficult to accept very long time horizons as meaningful because it would imply a certainty that trees will still be alive and hold carbon even in those very distant times. Apart from uncertainties with regards to fire, pests and changed societal attitudes, possible climate change in many regions could well lead to changes in growing conditions for forests that will render the growth and survival of trees uncertain in decades to centuries from now (e.g. Kirschbaum *et al*., 1996).

7 Wood products

Wood that is harvested and converted into wood products extends the effective carbon sequestration and storage life of carbon. It thereby extends the useful role that forests can play in storing carbon and preventing its emission to the atmosphere.

Data compiled by Winjum *et al*. (1998) (Table 16.3) suggest that in 1990, wood harvested globally contained 1120 MtC. Of that, 170 MtC was left on the

Table 16.3. Estimated fluxes (MtC $year^{-1}$) of carbon in forest operation and into wood products in 1990 (data from Winjum *et al.*, 1998). The estimate of the amount of the current pool of long-lived wood products has been taken from Matthews *et al.* (1996).

Total wood harvest	1120
Slash during harvesting	170
Fuelwood and charcoal	515
Harvested for commodity uses	436
Wastage during production	90
Commodity wood production	346
Short-lived products (< 5 years)	94
Long-lived products (> 5 years)	252
Emissions from decay of old wood	113
Sequestered in increasing pools of wood products	139
Total carbon emission from forestry operations	980
Total pool of wood products	2–8 GtC

forest floor as slash and 950 MtC was used productively. Of that, 515 MtC was used as fuelwood or charcoal and 436 MtC as industrial roundwood. A further 90 MtC was lost as wastage during industrial production. Some of that could be retrieved for additional fuelwood use.

A total of 346 MtC was used for the production of different commodities, sawnwood, woodbase panels, other industrial roundwood and paper and paperboard. Winjum *et al.* (1998) assumed that a specific fraction of each of these materials had short lifetimes of less than 5 years and the remainder longer lifetimes. This fraction assumed to have longer lifetimes included assumptions about the slow breakdown of wood and paper under anaerobic conditions in landfills. Decay rates for each product group were further broken down by latitudinal zone, with faster breakdown in warmer climates.

Using the fractions of fast and slow breakdown, Winjum *et al.* (1998) calculated that 94 MtC was produced in products with a lifetime of less than 5 years, and 252 MtC was produced in products with a lifetime longer than 5 years (including the slow decay in landfills).

These 252 MtC add to an increasing pool of wood and paper products maintained within society. However, even structural timber does not have infinite longevity, but is discarded or slowly decays. The total amount of carbon released during this breakdown is a function of the size of the total pool of wood and paper in use, and not strongly affected by a particular year's production. The carbon loss from the decay of these long-lived products was estimated as 113 MtC year^{-1} in 1990, which resulted in an estimated increase in the global stock of wood products by 139 MtC year^{-1}. This is an important contribution to global carbon stocks and amounts to about 2.5% of the amount emitted into the atmosphere from fossil fuel. It also partly accounts for the 'missing sink' in the total global carbon budget.

The total amount of carbon emitted through forestry operations is the amount harvested (1120 MtC) minus the amount sequestered through increasing the pool of wood and paper products (139 MtC) leaving an estimated 980 MtC, which compares to about 5.5 GtC year^{-1} released by fossil fuel burning (Fig. 16.1; Table 16.1). However, unlike fossil fuels, any carbon release from wood utilization can be largely balanced by regrowth provided forests are managed on a sustainable basis.

8 Wood use to substitute for fossil fuels

While vegetation sinks can play a useful role in mitigating against global warming, their potential role is also limited. Once a site has attained its maximum storage potential, carbon stocks stored on a site cannot be increased any further. Conversely, when all carbon has been lost from a site, no further carbon loss is possible, either. The amount of carbon contained on an area of land can vary only between zero and some site-specific upper limit.

If, at some future time, more urgent action to sequester carbon from the atmosphere is seen to be warranted, the option of planting trees to take up additional CO_2 would no longer be available on land that is already fully stocked with trees. Furthermore, if climatic changes were to render growing conditions for trees significantly worse, especially if warmer conditions were to increase the danger of forest fires, it could potentially lead to the release of large quantities of CO_2 (King and Neilson, 1992; Neilson, 1993; Smith and Shugart, 1993, Kirschbaum *et al.*, 1996). The more trees are planted now, the greater this potential liability becomes.

Hence, we should be cautious with using tree planting to reduce net Greenhouse gas emissions. Reducing net emissions by improved energy efficiency or reduced usage of fossil fuels does not suffer from these drawbacks. At the same time, forests can make a sustainable contribution to reducing net Greenhouse gas emissions by substituting wood for fossil fuels, either as an energy source, or by substituting wood for other materials, such as steel, aluminium or concrete that release larger amounts of CO_2 in their manufacture.

9 Wood as an energy source

Globally, biomass is currently the most significant non-fossil source of energy (Fig. 16.7). Biomass supplies about 14% of global energy needs (Nakicenovic *et al.*, 1996) of which about 75% are used in developing countries (Hall, 1997). A number of developed countries, especially the densely forested countries in northern Europe, also use it heavily (Table 16.4). Wood is the biofuel that is most widely used, but non-renewable peat also constitutes a significant source.

Currently, the use of biofuels is largely restricted to space heating in developed countries and cooking in developing countries. Biofuels are also used by some industries, especially those that utilize large amounts of biomass as an industrial raw material. Hence, the pulp and paper and sugar-processing industries are currently the main industrial users of biofuels.

Newer technologies, especially in wood gasification, have made electricity production from wood more cost-effective and energy-efficient. Other technologies, such as liquification of biofuels to produce transport fuels, have concentrated more on rapeseed oil and ethanol production from sugar rather than the use of wood. Wood is mainly used for heat production either for space heating or, indirectly, for electricity generation.

Detailed analyses have suggested that biomass has as much potential to contribute to renewable forms of energy generation in the near future as all other forms of renewable energy combined (Fig. 16.8). Some forms of renewable energy, such as solar, may have increasing potential in the more distant future, but rely on further technological advances before that potential can be fully met. The utilization of wood requires fewer of such technological advances, although they can further increase the attractiveness of wood. The current use of wood as biofuel is mainly constrained by costs which are partly increased by competition for land to produce food, and by competition for wood for other end-uses.

The other renewable energy sources that have potential to supply additional energy, such as wind and hydro energy, are restricted by the number of suitable sites, especially for hydro energy, and hence their potential for expansion is limited (Fig. 16.8).

Much of the expansion potential of biofuels derives from wood that is cut for a variety of reasons but not utilized further. In Australia, for example, a large area of forests and woodlands is cleared annually for the expansion or fostering of agricultural activity. A small fraction of the cut wood is used for

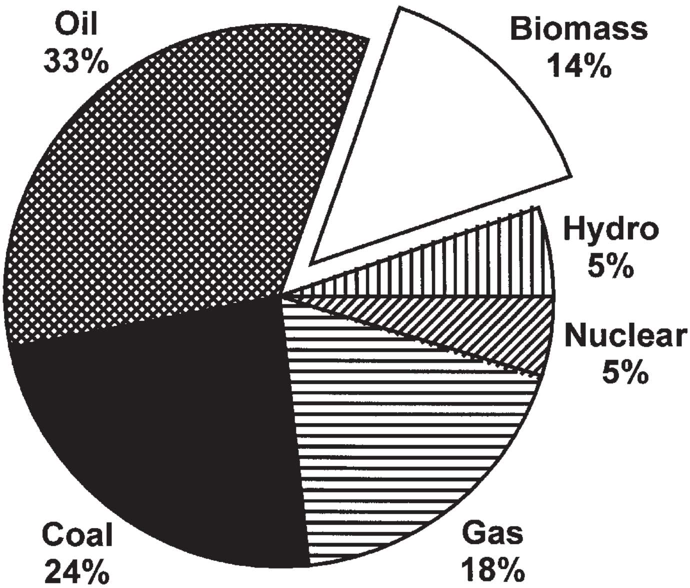

Fig. 16.7. Global energy consumption (1990) by energy source. Total energy consumption was 385 EJ year^{-1}. Data compiled by Nakicenovic *et al.* (1996).

Table 16.4. Fraction of energy requirements met by biomass in selected countries.

Finland	18%
Sweden	16%
Austria	13%
USA	4%
Australia	5%

Data for Australia are based on NGGI (1997a). Other data after Hall (1997).

energy generation, but the majority is not utilized and either burnt or left to decay on site. Figures compiled for Australia's National Greenhouse Gas Inventory show that in 1990, 19 MtC year^{-1} were cut in clearing operations and not further utilized (NGGI, 1997b). This resource could alternatively be collected and used for energy production. Assuming a conversion efficiency of 7.5 MJ (kgC)$^{-1}$ (P.Y.H. Fung, personal communication, CSIRO Forestry and Forest Products, 1999), this could be used to generate 570 PJ year^{-1} of energy and meet 14% of Australia's total energy needs. However, this would only be practically feasible where generating plants exist, or could be constructed, in the vicinity of the sites where biomass is available so that transport costs (and associated energy use) can be kept to a minimum.

10 Wood as an alternative to high-energy-requiring materials

All materials used in building construction consume considerable amounts of energy in their production, leading to significant fossil fuel emissions to satisfy these energy requirements (Table 16.5). Wood leads to the lowest emissions, as it requires only minor energy inputs in harvesting and sawing.

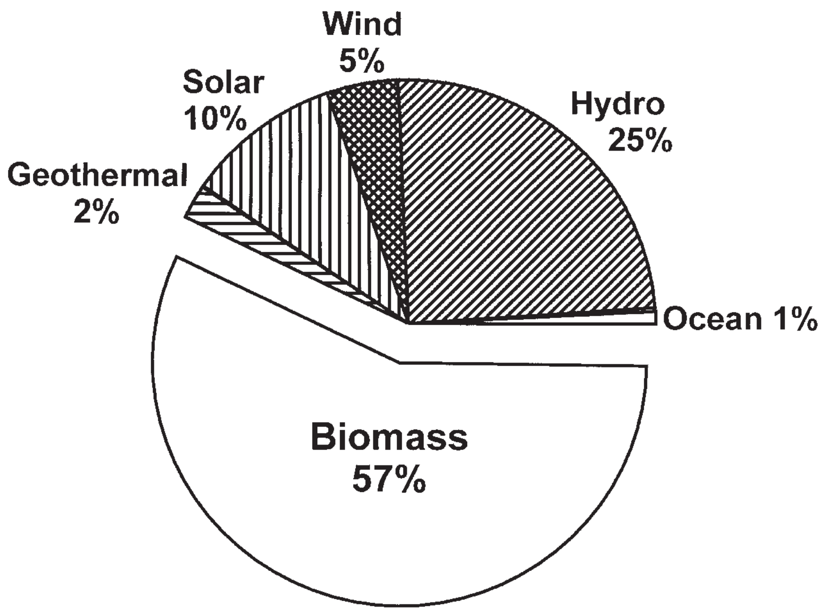

Fig. 16.8. Estimated potential contribution from different renewable energy sources by 2025. Total potential contribution is estimated to be between 130–230 EJ year^{-1} primary energy. Data compiled by Nakicenovic *et al.* (1996).

Table 16.5. Energy requirements for producing alternative building materials (after Ambrose and Tucker, 1996; Schopfhauser, 1998).

Material	Energy ($MJ\ kg^{-1}$)	C emission ($kgC\ kg^{-1}$)
Wood	2.4–6.0	0.06–0.15
Paper products	29	0.7
Clay products	5–7	0.12–0.17
Plastics	65–110	1.6–2.7
Steel	25–37	0.6–0.9
Stainless steel	450	11
Aluminium	182–187	4.5–4.6
Copper and alloys	83–107	2.0–2.6
Cement	6.1	0.29

Where a range of values is given, they reflect different estimates by the different literature sources. The value for cement is from Kashiwagi *et al.* (1996) and includes the non-energy related CO_2 release during cement manufacture. Conversion of energy to carbon emissions used the Australian average CO_2 release of 25 $kgC\ GJ^{-1}$ (Ambrose and Tucker, 1996).

Other typical industrial materials, such as steel, plastics and aluminium, on the other hand, have much greater energy requirements for mining, processing, smelting and, with some materials, reduction of oxidized ore. These energy requirements translate into corresponding CO_2 emissions although the absolute amounts also depend on the energy source that is used. Electricity generated from brown coal, for example, emits almost twice as much CO_2 per unit of energy produced as electricity generated from natural gas (Ambrose and Tucker, 1996), and wind, solar or hydro electricity does not directly generate any CO_2 emission.

Hence, any substitution of wood for any of the alternative materials listed in Table 16.5 could reduce energy requirements and associated greenhouse gas emissions. Schopfhauser (1998), for example, estimated that for a typical German house, more than half a ton of carbon could be saved per ton of extra wood used in construction. Half a ton of carbon would additionally be sequestered in the wood while it is structurally contained within the house. What makes the saving from material substitution particularly attractive is the fact that the saving is permanent whereas the carbon stored in building material is only temporary because of the eventual destruction of the material.

11 Measurements and verification

If carbon stored in forests is to be included in managing atmospheric levels of CO_2, such as under the Framework Convention on Climate Change, or more specifically the Kyoto Protocol, then incentives must be provided for countries or individual landholders to maximize carbon stocks on land under their control.

In general, such incentives would need to be linked to measurable carbon stocks, or changes in stocks, in the field. These measurements must be certified by generally accepted means. Different issues arise for large-scale measurements, such as for forest districts, regions or whole countries, than for measurements on individual properties, as might be required for individual carbon offset projects and carbon trading schemes.

Unfortunately, it is very difficult to assess whether any region is gaining or losing carbon because changes in carbon stocks are typically only small fractions of total carbon stocks, because forest growth and mortality are greatly affected by year-to-year variability and because many forests display a high degree of heterogeneity. Hence, measurements may be biased by being conducted in atypical years, on atypical material, or over such short time intervals that long-term trends are not apparent. Even after having established a trend for an area, it is generally necessary to know whether the trend has occurred under atypical climatic or disturbance conditions and may change once the area experiences more normal conditions.

These difficulties can be most effectively dealt with by an integrated assessment system making use of a variety of different measurement techniques that can mutually constrain and reinforce each other (e.g. Green *et al*., 1996). To assess carbon-stock changes cost-effectively over large areas, it is necessary that most relevant information be collected remotely. A monitoring system could have the following components.

11.1 Remote sensing

Remote sensing provides the only feasible method to obtain information on forest biomass and stand conditions over large areas in a timely and cost-effective manner. There is a range of available means for observing different attributes of forests, and additional methods are under development.

Aerial photography

Aerial photography has been used for at least 50 years for 'remote sensing' (Pitt *et al*., 1997). In the past, it has been mainly used for classifying forests by type or to distinguish between forested and non-forested land. One of the attractions of this technique is the historical perspective that can sometimes be obtained if older images are available (e.g. Fensham *et al*., 1998). Images can be obtained in stereo to allow fuller interpretation, and modern images can be obtained at different wave lengths and analysed electronically. Resolution depends on camera equipment and flying height, with greater resolution being at the expense of reduced coverage. Aerial photography is mainly suitable for local to regional-scale analyses.

Visible and near-infrared satellite imagery

Satellite images have been available for about 20 years. Images, such as Landsat, are obtained frequently and their coverage is global. Information can be purchased at a cost that may be high if only small areas are studied, but low compared to alternative methods if broad coverage, such as across a nation, is required. Satellite information is most useful for broad-scale assessment of the presence or absence of certain vegetation types, such as for an assessment of land clearance rates (e.g. Skole and Tucker, 1993).

Data are usually presented as the Normalized Difference Vegetation Index (NDVI), which is essentially a measure of projected leaf surface area (Sellers, 1985, 1987). NDVI information can be used directly as an input into forest growth models as a surrogate for local measurements of light interception (e.g. Coops *et al*., 1998). Data are available at different resolutions, with older data with coarser resolution being generally cheaper or available free of charge, whereas newer images with finer resolution are often more costly.

Laser altimetry

All optical methods, including aerial photography and satellite imagery, are essentially observations of leaf area development because that is the forest component that most strongly reflects or absorbs visible and near-infrared radiation. While total forest biomass is sometimes correlated with leaf area development, there are many situations where it is not.

Laser altimetry and radar-based measurements can overcome some of these problems. In laser altimetry, a light pulse is sent from an aircraft to the ground. By measuring the time it takes for the light pulse to be returned to the instrument it is possible to calculate the distance from the aircraft to the ground. Some light is reflected from the canopy and some passes through the canopy and is reflected back from the ground. This provides precise information of a region's ground elevation and average canopy height. Standing biomass can then be computed from correlations between canopy height and tree biomass. Laser altimetry is currently restricted to flying altitudes of less than about 1500 m (Pitt *et al*., 1997).

Synthetic aperture radar

Like laser altimetry, radar sensors emit their own electromagnetic radiation and record and deduce information from the backscattered signal. Radar detectors can be mounted on satellites such as the Japanese Earth Resource Satellite (Pitt *et al.*, 1997). Radar can be transmitted at different wavelengths and at different angles to the vegetation, with longer wavelengths being preferentially backscattered by larger tree components such as trunks and shorter wavelengths by smaller components such as leaves and branches. Radar measurements can be useful for detecting biomass amounts of up to 100 tDM ha^{-1} (Kasischke *et al.*, 1997). However, signals are usually noisy, as radar backscattering is also affected by other factors, such as slope, moisture conditions and tree phenology.

11.2 Forest inventories and harvesting records

The heterogeneity of landscapes and forest communities generally causes the correlations between remotely obtained information and actual biomass to hold only over regions with common characteristics. Hence, correlations between remotely sensed information must be based on ground observations in individual regions that share common landscape and vegetation attributes.

Information from forest inventories and harvesting records can provide such ground-based estimates of biomass. This information is required both as a direct check of biomass production simulated at a particular site and as direct input into forest growth models. The growth potential of forests differs greatly with forest age, and to simulate the growth of forests it must be known whether forests are young and newly established, or older forests whose growth has started to decline.

11.3 Growth modelling

In order to interpolate between times of measurement, to generate spatial coverage from point measurements and to assess long-term growth trends as affected by site and climatic characteristics, a forest growth model such as 3-PG (Landsberg and Waring, 1997), Promod (Battaglia and Sands, 1997), Forest-BGC (Running and Coughlan, 1988; Running and Gower, 1991) or CenW (Kirschbaum, 1999) is necessary. Such a model should be run with fine spatial resolution so that carbon accumulation for each site can be calculated within realistic constraints.

The growth model should explicitly include all relevant pools, and the size of pools should be modified through running the model. Rates of gas exchange should be driven by the size of existing pools, such as foliage carbon and nitrogen pools. Relevant cycles should be closed where possible, ensuring conservation of phosphorus and other nutrients and realistic constraints on water, nitrogen and carbon cycles.

Information about soils, topography and vegetation cover should be explicitly included as drivers of the growth model. Vegetation type is one of the key variables that determines the rate and time course over which vegetation can absorb or emit carbon. Where possible, available data for current standing biomass obtained from forest inventories should be used as a key modifier of future growth rates.

The models should be driven by information about soils, terrain attributes and observed weather and atmospheric CO_2 concentration, and be capable of predicting future sources and sinks under given assumptions of climatic and atmospheric change, land use, fire management and other management/harvesting strategies.

11.4 Atmospheric measurements

As an independent further constraint, atmospheric measurements of CO_2 concentration together with fluxes of other trace gases and heat can provide estimates of gas exchange rates from local to continental scales (e.g. Baldocchi *et al.*, 1996; Rayner *et al.*, 1996). Isotopic trace-gas composition can be used as an additional constraint (e.g. Enting *et al.*, 1995). These approaches set limits to rates of carbon exchange between the biosphere and the atmosphere, and thus provide an independent check on other methods of estimation. Atmospheric measurements provide integrated net fluxes over specified areas. Such approaches can help to overcome the difficulties that are inherent in spot measurements because of the large heterogeneity of the biosphere. These techniques are still in their development phase, but may become useful additional tools in the future.

12 Conclusions

As a large amount of carbon is contained in the world's forests, any change in these storage pools can significantly affect the amount of carbon in the atmosphere. Managing the world's forests for their contribution to the global carbon cycle is therefore warranted. The amount of carbon in forests can be maximized either through planting new forests or preserving existing forests with high standing biomass.

All these options relate to manipulating the size of a reservoir that is of strictly limited size. The greatest and ongoing contribution that forests can make to the global carbon cycle is through the substitution of wood for fossil fuels, either directly in energy production, or through the substitution of wood for high-energy-requiring alternative materials, such as aluminium.

There are no simple indicators by which to judge whether forests are maintaining their contribution to global carbon cycles. Potential changes in forest carbon stocks that would constitute significant changes in terms of the global cycle represent only small fractions of the total amount of carbon contained within forests. To detect such changes is difficult, and it is advocated that it can best be achieved through a combination of methods, including growth modelling, remote sensing, ground measurements and atmospheric trace gas studies.

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Ecosystem-level Forest Biodiversity and Sustainability Assessments for Forest Management

17

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Most discussion of biodiversity focuses on species, but an increased emphasis on the ecosystem level could bring considerable benefits for biodiversity conservation and its assessment in managed forests. Ecosystems are most usefully described according to their species composition, structure and physical environment, including disturbance regime: *ecosystem* therefore becomes a synonym of *forest type*. Ecosystem biodiversity is the number, variety and spatial arrangement of different forest types at a given scale, within the immediately superior hierarchical level of scale; the example of neotropical lowland moist forests shows it to be ubiquitous, arising due to varied causes. Sustainable forest management must be based on understanding of ecological differences between forest types in forest management units (FMUs), and the conservation of representative samples of forest types should be a management objective, thus conserving not only ecosystems, but also – through the 'coarse filter' principle – most of the species which make them up. Criteria and indicators (C&I) sets usually make some reference to ecosystem-level biodiversity and its evaluation, though the emphasis on this aspect in some sets seems insufficient. We suggest that the focus of ecosystem-level biodiversity C&I should run from the conservation of forest cover and the patterns formed by forest types in landscapes, to evaluation of the proportion of the area of each forest type within the FMU which is modified, and how, by management. Where necessary, programmes for the development of national or regional vegetation classifications could

make an important contribution to capacity for the assessment of sustainable management and biodiversity conservation in FMUs.

1 Introduction

The future of much of the biological diversity in the world's forests depends upon the way in which timber production forests are managed (International Tropical Timber Organization, ITTO, 1999a). This relatively recently-defined axiom of forest management is encapsulated by The Forest Stewardship Council's Principles and Criteria for Forest Stewardship (FSC, 1996), which require that forest management maintain intact, enhance or restore biodiversity at all its levels, including that of 'unique and fragile ecosystems'. What are the characteristics of biodiversity at the ecosystem level, what is the basis of the requirement that forest management conserve biodiversity at this level, and how might the fulfilment of this requirement be assessed through C&I? The general objective of this chapter is to provide a synthesis of concepts related to ecosystem-level biodiversity, its importance to sustainability and its characterization, value and assessment, through C&I, in the context of forest management. To illustrate forest biodiversity at the ecosystem level, information on neotropical lowland moist forests will be used. This is not a chapter on neotropical forests, therefore, but a chapter on ecosystem-level forest biodiversity – its meaning, importance and measurement, in which certain ideas are illustrated using neotropical forests as examples. Temperate and boreal forests may be better known from both ecological and management points of view, but the use of information from the neotropics may contribute to redressing the balance in this respect, which could be timely given the global importance of neotropical forests. In particular, much work on the ecology and management of temperate and boreal forests has been carried out in the globally extensive situations in which fire is a major ecological factor (Spurr and Barnes, 1980, cited by Attiwill, 1994), and/or in which severe forest disturbance may occur due to insect herbivores, and in which clear-cutting may be a widely employed management option for timber production (Attiwill, 1994). None of these situations pertains in the moist neotropics, so that the consideration of this region may also contribute to broadening the scope of understanding and applicability of C&I.

In the light of the preceding comments, this chapter has the following structure. Firstly, current concepts of ecosystem-level biodiversity and its measurement, in both research and practical contexts, are reviewed, and we suggest that ecosystem-level C&I should play an important, or even central, role in the assessment of the sustainability of management and the conservation of biodiversity at the FMU level. Then the nature of ecosystem-level forest biodiversity, and the historical, physical, biological and human factors which underlie it, are illustrated through a synthesis of information concerning forest of the moist neotropical lowlands. The final section reviews the approaches taken to ecosystem-level evaluations of forest resources in current C&I processes, compiles a possible set of C&I for biodiversity assessments at the ecosystem level in FMUs and suggests some future directions for research and development related to C&I.

2 Ecosystem-level biodiversity: principles

2.1 Biodiversity and ecosystems

Although definitions of the term biodiversity[1] may vary in their wording, there is a clear consensus as to what the term and concept embrace – the variability or diversity among living organisms within species and between species, and of the ecological complexes, including ecosystems, of which the species are part (e.g. McNeeley *et al.*, 1990; World Conservation Monitoring Centre, WCMC, 1992; Boyle and Sayer, 1995; Harper and Hawksworth, 1995; Heywood *et al.*, 1995; Gaston, 1996a). Many discussions of biodiversity focus, whether explicitly or implicitly, on its compositional or taxonomic aspect – the particular genes, species or types of natural community (ecosystem) present in a sample. Noss (1990) and Harper and Hawksworth (1995) remind us, however, that the variety of life includes not only compositional, but also structural and functional aspects. The structural aspects of an ecosystem may be thought of as the physical spatial pattern of the objects which form it, such as the numbers of trees in different size classes or the spatial patterns which natural communities form over landscapes; function involves processes such as gene flow, and cycles and fluxes of matter and energy (Noss, 1990).

Importantly from the point of view of biodiversity assessments using C&I, the term ecosystem has come to have both a strict and a ‘liberal’ (Noss, 1996) usage. Strictly, it refers to a unit formed by both living and abiotic components (e.g. vegetation and soil) in reciprocal interaction; in the modern study of ecosystems the interactions of interest are the flows and cycles of energy and matter which are part of the functioning of the ecosystem (Waring, 1989). A more liberal usage of the term is appropriate in applied contexts such as that of the present chapter, and is exemplified by the definition of an ecosystem as ‘a physical habitat with an associated assemblage of interacting organisms’ (Noss, 1996). FSC (1996) also adopts this liberal usage, in which the term ecosystem may be interpreted as a simple synonym of others such as natural community or vegetation type, understanding community as an assemblage of species populations which occur together in space and time, often under a particular set of environmental conditions (Begon *et al.*, 1996). It is in this liberal sense, of course, that the term ecosystem is used in the context of assessments of sustainability and biodiversity conservation for forest management, and this is the sense that is used henceforth in the present chapter. While the liberal usage of the ecosystem concept is what makes it operational, strictness must be retained with respect to the spatial scale under consideration, which must always be stated. This is because, as Noss (1996) points out, the definition of the term ecosystem offered above potentially encompasses a wide range of spatial scales; although the most frequent scale at which ecosystem diversity is assessed is probably that of forest types within landscapes, it may be detected at scales from the local to the regional, as is shown later.

The differences between ecosystems – of the species composition of their biotic components, of their structure in terms of parameters such as biomass, and of their functioning in aspects such as biomass productivity – are one of the most obvious and important facets of the biosphere. The concept of ecosystem biodiversity (defined formally below) is, to a great extent, simply a novel way of unifying all the aforementioned and long-recognized aspects of the differences between ecosystems. In terms of their biotic components, ecosystems are biological entities definable not only in terms of their composition and structure, but also by interactions between species and functional properties. Ecosystems are not islands, however – energy, materials, organisms and genes move between them across landscapes (defining the term landscape, following Forman and Godron (1981, cited by Noss, 1983) as a 'kilometers-wide area where a cluster of interacting stands or ecosystems is repeated in similar form'). Understanding of ecological phenomena which operate at the scales of the landscape and region is now recognized as vitally important to the sustainability of land management (Odum, 1992; Forman, 1993) and the widely referred-to spatial scale of the community type within the landscape links the relatively well-understood level of species biodiversity to these much less well-understood higher levels (Noss, 1983; Franklin and Forman, 1987; Lapin and Barnes, 1995).

For assessments of biodiversity as well as for other purposes, different forest ecosystems may potentially be delimited on the basis of any or all of the criteria mentioned above – compositional, structural and functional. Functional criteria, however, are costly and time-consuming to measure and are certainly inappropriate, on their own, for biodiversity assessments, as forest types of different taxonomic composition may have similar functional properties. Structural criteria are informative and relatively easy to measure, but as in the case of functional aspects, often lack resolution for the detection of forest types when used alone, except, for example, when comparing early and late stages of forest recovery after drastic disturbance (many of the authors cited in this chapter's section on ecosystem diversity in neotropical forests comment on how the apparent physiognomic and structural uniformity of neotropical moist forests conceals considerable compositional variation at multiple scales). The most practical way to approach the evaluation of ecosystem biodiversity in forests is that which ecologists have traditionally used for the description of forest types – on the basis of compositional and structural criteria, and the environmental conditions, including disturbance regimes, under which forests of certain structural and floristic characteristics tend to occur (see later sections).

2.2 Obtaining, understanding and using information on diversity and biodiversity: easy for species, difficult for ecosystems?

Some authors have argued that biodiversity of ecosystems is a more difficult concept to understand, communicate and apply to resource management than biodiversity of species (Boyle and Sayer, 1995; Gaston, 1996b). Species counts have become the most widely and frequently applied measure of biodiversity (Wilson, 1988; Boyle and Sayer, 1995; Harper and Hawksworth, 1995; May, 1995; Gaston, 1996b) and the level of understanding of biodiversity at this level greatly exceeds that of ecosystem-level biodiversity. Study of a selection of recent treatises on biodiversity, for example, shows that in most cases, they either do not treat the ecosystem level (e.g. the papers edited by Wilson and Peter, 1988; Hawskworth, 1995; Gaston, 1996c; Reaka-Kudla *et al.*, 1997) or that they reach it only at the broadest (global) scale possible (WCMC, 1992). Only the Global Biodiversity Assessment (Heywood and Watson, 1995) attempts comprehensive coverage of basic concepts related to biodiversity at the ecosystem level. In spite of these precedents, we will here attempt to show that the ecosystem level becomes tractable and valuable, from the operational point of view, if some simple and important assumptions are accepted.

Ecologists have usually studied diversity at the species level and may measure it as either the number (or *richness*) of species in the community or area under consideration, or in terms of an index of diversity (Greig-Smith, 1983; Pielou, 1995). Diversity indices are generally calculated on the basis of species richness and the *evenness* of the contribution of the different species to the community. Values of diversity indices therefore vary in relation to both the number of species in a community and the evenness of their proportional abundances, with the highest diversity found where proportional abundances vary least, i.e. evenness is greatest.

Much less attention has been paid to the study of the diversity and biodiversity of ecosystems, but the same principles of measurement apply at this higher level. The simplest and most appropriate way to define the diversity of forest ecosystems is as the number, variety and spatial arrangement of forest types at a given scale (see later), within a given study area. Especially in the context of FMUs, the most useful and usual scale will probably be that of forest types defined at local scales, within landscapes (Romme, 1982, see fig. 1; Burke *et al.*, 1995; Hengeveld *et al.*, 1995; Lapin and Barnes, 1995). In the same way as species diversity, ecosystem diversity may most easily be measured as richness – the number of different ecosystems in a landscape – or by the calculation of diversity indices. For the calculation of diversity indices, the proportion of land area occupied by each ecosystem may be used as the measure of its contribution to landscape cover, and other factors such as the degree of interspersedness of the different ecosystems in a landscape may be taken into account (Romme, 1982; Fig. 17.1). In parallel with Harper and Hawksworth's (1995) comments on biodiversity at the species level, evaluations of ecosystem biodiversity should ideally take into account not only how many different ecosystems exist in an area, but also, how different they are from each other (Whittaker's (1975), p. 119) concept of β-diversity measures was that they should express the degree to which communities differ from each other because of separation along environmental gradients). Finally, the study of ecological diversity is value-neutral, but biodiversity evaluations must often weight taxa, for example on the basis of their commonness or rarity (Pielou,

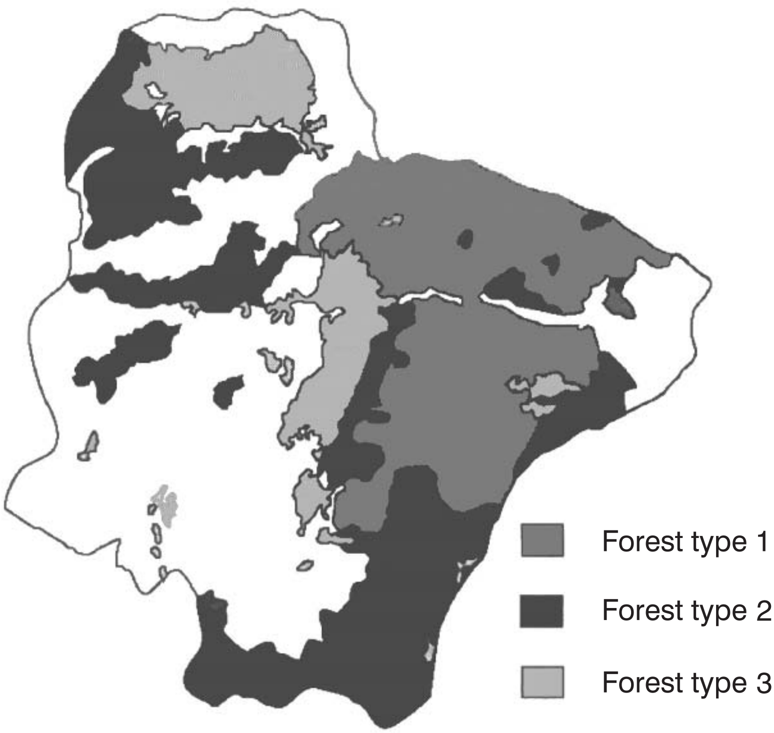

Fig. 17.1. Ecosystem diversity in a landscape as illustrated by Romme's (1982) reconstruction of the forest types (defined by floristic and structural characteristics) of a 73-km^2 watershed in Yellowstone National Park, USA. The map shows three forest types as examples, and measurement of the diversity of ecosystems in the landscape can be based on the number of different forest types and the proportions of the total landscape area occupied by each one (see text; redrawn from Romme, 1982).

1995): at the ecosystem level, weighting could be carried out in relation to factors such as the areal extent of different forest types, the numbers or proportions of endemic species known to occur in them, the degree of threat to ecosystem integrity and the area under protection.

If ecosystem diversity is to be measured, of course, then we must be able to identify and delimit ecosystems. The description of communities has been a basic activity of ecological science since its inception and some of the classic studies of forest community ecology, such as that of forest types which make up the landscape of the Great Smokey Mountains of Tennessee, USA (Whittaker, 1956, cited by Begon *et al.*, 1996) may now be thought of as studies of ecosystem biodiversity. The scientific literature on the identification, description and comparison of plant communities is extensive (e.g. Greig-Smith, 1983; Magurran, 1988), though a great number of alternative methods for the classification of ecosystems exist and different criteria may be used, with different results (Hengeveld *et al.*, 1995). Whatever methods are used, however, it remains the case that when characterized in terms of plant species composition, there are often no clearly defined limits or boundaries between two adjacent ecosystems; in principle, then, ecosystems are much less well-defined than species as units for the measurement of diversity or biodiversity (Noss, 1987; Scott *et al.*, 1991; WCMC, 1992; Boyle and Sayer, 1995; Hengeveld *et al.*, 1995). Ecosystems do not have clearly defined limits for many reasons. Plant species may typically be present across a wide range of habitat conditions in a forest landscape, their abundances vary markedly across that range and are affected not only by the physical environment, but also by historical factors, disturbance and biotic interactions, and their behaviour is strongly individualistic – that is, each species shows an essentially unique response, in terms of its distribution and abundance, to the factors which shape the characteristics of the vegetation (Begon *et al.*, 1996, Chapter 16). Thus while we may compare the vegetation of a hilltop with that of a valley bottom and find that they differ markedly in their composition, if we sample the vegetation between the hilltop and the valley bottom, we will often find a gradual transition, not a sharp boundary, between the two. If it is necessary to establish limits of vegetation types within such a continuum – for example, in the development of a classification of forest types for land management purposes – such limits will necessarily be arbitrary. Besides the preceding factors, we must take into account that ecosystems lack clearly defined limits in time as well as in space. Ecosystem composition changes over time as the distributions and abundances of the constituent species populations change in response to multiple factors, of which climate change is perhaps one of the most important; even in relatively undisturbed landscapes, the current species composition of a forest should be considered, in the long term, a transitory state of a continuously changing system (e.g. Hunter *et al.*, 1988; see later).

The implications of difficulties in the measurement, interpretation and application of the ecosystem biodiversity concept, such as those noted above, have been analysed in relation to forest management and conservation by several authors. Boyle and Sayer (1995) review the subject in the context of biodiversity conservation in managed tropical forests. They conclude that especially in comparison to species, ecosystems are not very distinct and that their biodiversity may be difficult to measure if multivariate statistical techniques are required. In other contexts, however, such worries about the delimitation of ecosystems have been robustly ignored. Bailey (1980) reviews systems for ecological land classification, on the premise that 'classification of land is required to provide an effective basis for resource assessment and management and land use planning'. Approaching a partial definition of ecosystem biodiversity, he observes that ecological land classification systems 'describe and define taxonomic ecosystems and ecosystem associations in relation to their geographic arrangement'. Reviewing site classification systems used in forestry, Kilian (1980) similarly expresses ideas that would later become part and parcel of the biodiversity concept, stating that the objective of such systems should not be merely the evaluation and prediction of land productivity in terms of increment, but 'the description of ecological facts, the differentiation of the various ecosystems in the landscape . . .'. Firmly in the biodiversity era, Noss (1987) shows how this approach has been adapted to the necessities of conservation biology, stating that management for conservation often requires plant community classifications and vegetation maps, which are important for the protection of species and ecological processes. He describes the 'applied community classification' approach used in the USA by The Nature Conservancy. In a later paper (Noss, 1996) he states 'as long as we realize that every classification or map is an abstraction invented for convenience, they can be tremendously useful', a point of view echoed by Hengeveld *et al.* (1995).

Knowledge of the characteristics of ecosystems and their distributions across landscapes and regions, then, is widely held to be a basic tool for natural resource management. The arbitrary delimitation of boundaries between ecosystems, while inconsistent with ecological reality, is a necessary step which should not detract from the operational value of the resulting information. Furthermore, it is a mistake to believe that forest ecosystems can be identified only by sophisticated research techniques. As Noss (1987, 1996) and others have argued, the considerable effort that basic ecological science has dedicated to the development of such techniques should not obscure the existence of much simpler approaches which can and should be adopted for land management purposes. Emphasis on practical approaches to the identification and mapping of ecosystems has accompanied one of the most significant trends in conservation biology in recent years – a move away from species-centred approaches to conservation action, to emphasis on the ecosystem level. This trend has occurred because of the enormity of the task of managing on the basis of detailed knowledge of individual species and their interactions, which has made the focus on the conservation of habitat – ecosystems and landscapes

– essential (Noss, 1987). The habitat conservation approach is exemplified by the ‘coarse filter’ strategy, developed in North America by The Nature Conservancy (Hunter, 1991; Noss, 1996). This strategy is based on the conservation of representative examples of the ecosystems of a landscape or region, both because these are biological entities in themselves and because the conservation of ecosystems ensures the conservation of a large proportion of the species they contain – without requiring information on those species. The small proportion of species which are not well served by this approach (which ‘fall through’ the coarse filter) must receive more detailed, individual attention from researchers and managers (the ‘fine filter’) (Hunter, 1991). The coarse filter strategy is not, of course, without problems, among which looms the instability of ecosystem composition over time (Danielson, 1994). The coarse filter strategy is complemented by the ‘gap analysis’ approach to the determination of priorities for conservation action (Scott *et al.*, 1991). A basic element of gap analysis is the identification of the degree of protection of the different ecosystems of a region. Ecosystems which are poorly represented in protected areas – ‘gaps’ in the coverage – may then be assigned priority for action (Scott *et al.*, 1991; Noss, 1999). An optimum gap-identification strategy, however, would be multi-faceted, taking into account the distributions of centres of species richness and endemism for different animal groups, as well as vegetation types (Scott *et al.*, 1991).

Major current conservation biological applications of the coarse filter/gap analysis approach are at scales considerably larger than those of most forest management units (e.g. Caicco *et al.*, 1995; Kiester *et al.*, 1996). To the degree that this approach optimizes biodiversity conservation in the face of insufficient information and the difficulties associated with species-centred approaches, however, it offers clear potential for adaptation to planning for biodiversity conservation in FMUs, and to sustainability assessments using C&I.

3 Ecosystem diversity and the factors which underlie it, illustrated by examples from neotropical lowland moist forest

3.1 Introduction

The intention of the present section is to demonstrate the reality and nature of ecosystem diversity in forests, as described and analysed from the plant ecological point of view. Factors that underlie ecosystem diversity and which are reviewed include forest history in relation to climate change, the geological evolution of the American tropics and the influence of pre-Columbian civilizations, as well as present-day relationships of vegetation to variation in the physical environment, disturbance and other factors, at different spatial scales. Although neotropical moist forests differ from temperate and boreal forests with respect to important factors such as types of natural disturbance (see Introduction), the types of ecosystem diversity discussed in the present section, and the factors which underlie them, are not exclusive to the neotropics. Thus the broad framework set out for the presentation, and the conclusions reached, may be considered relevant to forests in all climatic zones. As a final introductory comment, it is emphasized that the papers by Romme (1982) and Lapin and Barnes (1995) are prominent among the few studies which explicitly seek to characterize the diversity of a landscape in terms of its constituent ecosystems; the objectives of most studies, as will be seen below, are the identification of forest types and the factors which may underlie their differentiation and distributions. These studies demonstrate the existence of ecosystem diversity, without measuring it explicitly.

The broad approach used for the analysis of present-day forest floristic and structural variation is based on that of Whitmore (1984, 1990). He set out a 'rough hierarchy' of factors which underlie the distribution of different forest types, in order of diminishing importance. Biogeography is the first factor, which creates considerable floristic differences between the moist forest floras of different continents, for example, as well as regional variation within continents (see later). Next are different regimes of natural disturbance, which create predictable differences between forest types whatever their physical environment (see later). Whitmore then defines *forest formations* (another concept equivalent to that of the ecosystem, under liberal usage) on the basis of forest structural and physiognomic characteristics and the environmental conditions under which they typically occur. The first subdivision of environments which leads to the definition of forest formations is between ever-wet and seasonally dry macroclimates and the second, within macroclimatic regimes, is between dryland forest and forest on soil where the water table is high, at least periodically. Subdivisions are subsequently made by locality (inland or coastal), soil type and elevation, respectively. Soil types taken in by Whitmore's hierarchy include 'typical' or 'zonal' soils, and 'atypical' ones such as podzols and soils derived from limestone. This hierarchy takes us to broadly defined forest formations such as tropical lowland evergreen rainforest and freshwater swamp forest, in Whitmore's terminology. These forest formations are recognizable, on the basis of their structure and physiognomy (and in spite of the fact that their taxonomic composition varies widely), throughout the tropics. Forest formations are not homogeneous, however, different forest types typically occurring within them in relation to two sets of factors. The first is variation in substrate factors such as topography and soils at smaller scales than those which define the forest formations, which will be dealt with below. The second, which may be evident within areas of relatively homogeneous environmental conditions, is related to the particular regeneration patterns of individual tree species, which could potentially produce floristic variation from place to place which also shifts over time.

As discussed above, the degree and type of variation in vegetation composition is typically dependent on the spatial scale under consideration (Greig-Smith, 1983); we will explicitly consider regional, landscape and local scales in the sections following that on forest history. The landscape and local levels are the most directly relevant to the FMU, but the regional scale provides essential background information – for example, with respect to how unique or important a managed forest is in a 'broad context' (Noss, 1999). As a point of reference, the regional scale may be taken as referring to distances > 10^5 m, the landscape to the range 10^3–10^5 m and the local to the range 10–10^3 m (cf. Tuomisto *et al.*, 1995). In relation to scale, variation in forest composition may be thought of as nested: local-scale floristic variation related to topography may exist within a single forest type defined at the landscape scale in relation to broad categories of soil type.

3.2 Forest history: climate change, geological evolution and pre-Columbian civilization

The fact that the characteristics of natural communities change over time, and some of its implications, has been mentioned above. Among the major natural causes of temporal change in the composition of forests, and of floristic patterns observed at larger scales, are regional climatic and geological histories. The long-term geological and climatic dynamism of the American tropics is believed to have made a major contribution to the biodiversity of the region's forests, the highest of all the tropical regions. Three major factors shaping the broad modern patterns of neotropical forest biodiversity may be identified, two geological and one climatic (Gentry, 1982), and these are now discussed.

The current constitution of the American continent is relatively recent – in geological terms, of course. The existing continuous land connection between North and South America was established by the closing of the Central American isthmus between 3.1 and 2.8 million years BP (before the present) (Coates and Obando, 1996), though a period of biological interchange, perhaps along island arcs, had probably occurred at a much earlier time (see review by Gentry, 1982). The uplift of the Andes was the second major geological event of the Cenozoic, being completed only in the last 5 million years in the northern part of the range (Gentry, 1982). Major plant taxonomic differences between the communities of montane and lowland areas of the neotropics are probably due to the separation of North and South America during most of the Cenozoic, while the uplift of the northern Andes brought about major episodes of speciation in that region and in the lower Central American isthmus (Gentry, 1982). Finally, climate fluctuations during the Quaternary period, which began 2 million years ago, have been marked. The well-known refuge hypothesis seeks to link present-day distributions of areas of high endemism and species richness (a regional-level manifestation of ecosystem diversity) to the isolation of moist forest refuges amid areas of seasonal vegetation during glacial periods (e.g. the papers edited by Prance, 1982). This hypothesis is questioned by evidence that temperature reductions, not aridity, are the predominant climatic response in the wet tropics during glaciations (Colinvaux, 1996). In addition, the putative areas of high endemism and species richness may turn out to be artefacts of inadequate data. Whatever might be the outcome of these debates, it is evident that the composition of large areas of neotropical lowland forest has undergone constant long-term change during the Quaternary, and has contributed to current regional patterns of ecosystem diversity.

Humans have also been major players in forest history. Maya civilization, for example, occupied much of what are now the lowland tropical moist forests of Mexico, Belize, Guatemala and part of Honduras. Besides the fact that considerable areas must once have been cleared of forest and present-day vegetation is therefore secondary, it is believed that the present-day abundances of useful tree species such as *Brosimum alicastrum*, *Manilkara zapota* and *Spondias* spp. are due to these species having been favoured by Maya land management practices (Barrera *et al.*, 1977; Rico-Gray *et al.*, 1985). At the other end of Mesoamerica, the abundance of the light-demanding canopy tree species *Cavanillesia platanifolia* in present-day forests of the Darién region is suggested by Hartshorn (1980) to be due to pre-Columbian forest disturbance and clearance.

3.3 Ecosystem diversity, environmental variation and natural disturbance

Models of tropical moist forest ecosystem diversity

The explanation of the characteristics of ecological systems through simple models has always been an elusive goal. Early students of tropical forests (e.g. von Humboldt, cited by Richards, 1976) found their species diversity so overwhelming that discussion of forest composition and its variation was rarely attempted. The first half of the 20th century saw this mental block overcome and the birth in the 1930s of divergent points of view as to the underlying causes of tropical forest ecosystem diversity. These points of view are compared and contrasted by Richards (1976) – on the one hand, work by Davis and Richards in Guyana showed that local substrate variation may create predictable patterns of ecosystem diversity, while on the other, Aubreville's mosaic theory of regeneration postulated that ecosystem diversity would never be predictable in such terms, but depended on the spatially and temporally fluctuating regeneration patterns of individual forest plant species. Today it seems clear that both these points of view partly explain tropical moist forest ecosystem diversity, but single factors never account for even the greater part of the diversity encountered; indeed, Condit (1996) predicted that attempts to relate compositional variation of tropical forests to environmental factors will always find most of the variance in the error term. He favoured an updated version of Aubreville's theory, invoking chaos, for the explanation of this error term, and suggested an approach to the solution of these outstanding problems through basic research.

For the purposes of the present chapter, it is clearly necessary to review current understanding of the causes of differentiation of forest types in tropical moist forest – in other words, the factors which underlie ecosystem diversity – in more detail and, while emphasizing that knowledge is still very incomplete in some senses, extract the clearest conclusions possible regarding the application of this knowledge to C&I for sustainability.

The role of natural disturbance

In most environments, worldwide and including the moist tropics, biodiversity has evolved and is maintained in situations in which natural disturbance is an important and persistent factor (Pickett and Carpenter, 1995). In general, the evaluation of disturbance and its effects requires explicit characterization of the system under consideration, disturbance type, spatial scale and distribution, intensity and frequency, and a reference state (Pickett and Carpenter, 1995). Following Woodley and Forbes (1997) and Noss (1999, his table 1), however, we organize information in the present chapter with respect to a simple dichotomy of disturbance regimes. 'Gap' disturbance regimes are those in which the forest canopy is opened by the deaths of individual trees or small groups of them. Disturbances are therefore occasional and small-scale within the stand, and although they are important with respect to within-community species diversity (see above) (Denslow, 1987), they do not create ecosystem diversity because of their small scale. Ecosystem diversity in forests with gap disturbance regimes is more likely to be explicable in terms of variation in the physical environment, for example in substrate conditions. 'Stand-replacing' disturbance regimes are those in which forces such as hurricanes, fire or floodplain dynamics destroy whole forest stands and bring about their replacement by new ones. In contrast to gap regimes, stand destruction and replacement have an important direct effect on ecosystem diversity because of their intensity and larger scale, which create post-disturbance communities with a different structure and, often, species composition, from those which exist pre-disturbance, as well as from communities in the same landscape or region but disturbed at other times, or in which the disturbance regime is the gap type. Because of the different effects the two disturbance regimes have on ecosystem diversity, the gap/stand-replacing dichotomy is a useful device for the organization of information on this subject. It nevertheless belies the facts that disturbances and their effects form a continuum (Pickett and Carpenter, 1995), and that any forest landscape in which the disturbance regime is predominantly gap is likely to contain some patches recovering from stand-replacing disturbances (Whitmore, 1990; see below).

3.4 Ecosystem diversity in neotropical lowland moist forests with 'gap' disturbance regimes

The regional scale

Part of the regional variation of forest composition within forest formations may be explained in biogeographical terms. The tropical lowland moist forests of Ecuador, for example, are found in two regions separated by the Andean Cordillera: the north-western region in the Pacific coast lowlands, and the Amazon region (these forests were respectively designated as the Western Ecuador and Napo ecoregions by Dinerstein *et al.* (1995); see Fig. 17.2). Due possibly to the geographical isolation of the two ecoregions (this section), they differ markedly in their floristic characteristics. Few species are shared between the two forest regions, and the diversity of tree and liana species in 1.0-ha plots may be up to 50% greater in Amazon than in Pacific coast lowland forests, though the latter typically have a far greater representation of endemic species than is the norm in the Amazon basin (Palacios, unpublished data). The Pacific coast lowland forests are much more accessible than those of the Amazon basin and this fact, coupled with their high levels of species endemism, means they have much the higher priority for conservation action of the two ecoregions (Dinerstein *et al.*, 1995).

Other aspects of regional variation of forest composition within forest formations may also be biogeographical in origin, although the exact processes involved are less evident than in the case of geographical isolation. The broad community types differentiated in relation to substrate variation in Amazonian Ecuador are the same as those of the rest of the Amazon basin (see below), but some of the characteristic species found in them change along a north–south gradient (Palacios, unpublished data). The moist lowland forests of the Atlantic slope of Central America (united as the Central American Atlantic Moist Forest Ecoregion by Dinerstein *et al.* (1995); see Fig. 17.2) are dominated by the legume tree *Pentaclethra macroloba* up to a point in Nicaragua between the San Juan River and the city of Bluefields. Many of the canopy tree species which coexist with *Pentaclethra* up to this point are found right up to the north coast of Honduras, however (personal observations of the authors; Natural Forest Management Unit, CATIE, unpublished information), so that the division between *Pentaclethra* and non-*Pentaclethra* dominated forest may be considered a major floristic division of the ecoregion. Within northern Costa Rica, there is an approximately east–west gradient of forest composition, such that *Pentaclethra* is scarce in wet forests in the central part of that area of the country (Zamora, unpublished data). Finally, geographical longitude was one of the factors most closely related to compositional variation of dryland forest along a 200-km stretch of the middle Caquetá area of the Colombian Amazon (Duivenvoorden, 1995).

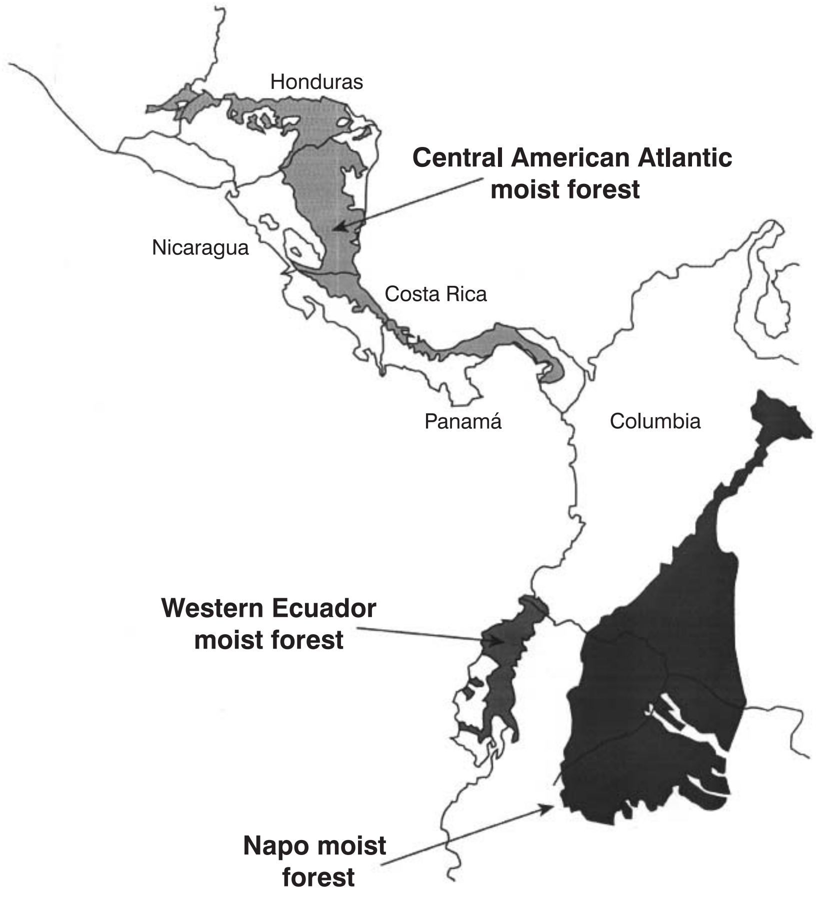

Fig. 17.2. Location of the neotropical terrestrial ecoregions mentioned in the text, redrawn from Dinerstein *et al.* (1995). The Central American Atlantic moist forest is bordered to the south and east by montane forests in Costa Rica and Panama, by dry forests in northern Costa Rica and southern Nicaragua and by pine forests in central and northern Nicaragua, Honduras, Belize and Guatemala. The two South American ecoregions are separated by Andean montane forests. The Western Ecuador moist forest is bordered on the Pacific coast by dry forests, and the Napo moist forest by other Amazon basin moist forest types. The south-eastern sector of this latter ecoregion is penetrated by the varzea (swamp) forest ecoregion associated with major rivers.

Besides biogeographical factors, large-scale variation in overall substrate conditions may also influence forest compositional variation at the regional level. Gentry and Ortiz (1993) review recent advances in knowledge of the flora of western Amazonia, asserting that significant differences of species composition exist between the forests of relatively fertile soils of the Andean piedmont (part of a band of relatively fertile soil which extends from Central America along the Andean piedmont south to Bolivia) and those of more infertile soils at greater distances from the Andes. In general terms, widely separated sites within this band of fertile soil may be more similar to each other than to closer sites on infertile soils; the flora of Cocha Cashu Field Station in Manu National Park, Perú, for example (relatively fertile piedmont soils), appears to be more similar to that of Barro Colorado Island, Panamá, than to that of the poorer soils of the Iquitos area, Perú (Gentry and Ortiz, 1993). Possible regional patterns in the degree of ecosystem diversity over landscapes, suggested by analysis of Landsat images, may also be linked to soil variation (Tuomisto *et al.*, 1995; this study is described more fully below).

Landscape and local levels – forest types of 'atypical' soils

At the landscape and local levels, the first two steps to be taken in the characterization of ecosystem diversity relate to substrate water regimes and the presence of 'typical' or 'atypical' soils (see above). Regarding substrate water regimes, we may separate dryland forest from forest on sites with a permanently or periodically high water table, which in inland freshwater sites may be generically termed swamp forest (Fanshawe, 1952; Encarnación, 1985; Pires and Prance, 1985; Whitmore, 1984, 1990; Tuomisto, 1993; Duivenvoorden, 1995). Tropics-wide, the comparison of swamp forest with adjacent dryland forest shows that the former has unique dominant species and lower within-community species diversity than the latter (Richards, 1976; Dumont *et al.*, 1990; Whitmore, 1990). Depending on local topography, swamp forest patches may be of only a few hectares in extent (see examples in Lieberman *et al.*, 1985, and Hubbell and Foster, 1986) while at the other extreme of scale, swamp forests (*varzea*) extend for hundreds of kilometres along the major rivers of the Amazon basin (e.g. Dinerstein *et al.*, 1995).

Probably the most important example of atypical dryland forest soils in the neotropics are the white sands, particularly associated with the Negro and Orinoco rivers (Whitmore, 1990) but found to a greater or lesser extent throughout Amazonia (Fanshawe, 1952; Encarnación, 1985; Pires and Prance, 1985; Tuomisto, 1993; Duivenvoorden, 1995). White sand soils and their characteristic vegetation are absent from the geologically recent terrain of Central America. Whitmore (1990) refers to this vegetation type as heath forest, a collective term whose usefulness is probably limited by its eurocentricity (it is derived from the German *Heidewald*; T.C. Whitmore, Department of Geography, University of Cambridge, July 1998, personal communication); numerous regional names for it exist in the neotropics. The unique characteristics of heath forest – low species richness, unique species and often but not always, lower canopy height than the surrounding forest – are thought to be related to periodic severe lack of water caused by the free drainage and poor water-retention capacity of the sandy soils (Jordan, 1985; Whitmore, 1990).

Landscape and local levels: forest types on 'typical' soils

After the relatively simple steps of separating forest types according to broadly defined substrate water regimes and the presence of atypical soils, there now remains the problem of characterizing floristic variation at landscape and local levels in the greater part of the forest area under consideration: dryland forests on 'typical' substrates. This is perhaps the least well-understood situation as far as vegetation–site relationships in neotropical moist forests are concerned (Clark *et al.*, 1998), though the information available certainly permits some tentative generalizations.

A study of potentially far-reaching implications for understanding of Amazonian biodiversity was recently reported on by Tuomisto *et al.* (1995), based on a variety of techniques including analysis of Landsat images covering 500,000 km^2 of western Amazonia. Their results indicate the existence, in Amazonian dryland forest landscapes, of a great diversity of floristically distinct patches which they called biotopes, with a mean length of 4.6 km. Forest areas covered by a given Landsat image (185 × 185 km) may include 30–40 biotopes within dryland vegetation, and the differences between forest areas in the number of biotopes are an indicator of regional-scale floristic variation, as mentioned above. Preliminary evidence indicates that the existence of distinct biotopes is linked to substrate variation, and it is possible that these links are general, especially given the enormous potential variation of substrate conditions in the western Amazon floodplain (Tuomisto *et al.*, 1995). These authors observe that maps of vegetation and soil properties are among the spatial data needed for precise hypothesis formulation and testing in Amazonian ecology and biogeography, and as a basis for the conservation of representative habitats (see earlier) in the region.

Several studies have established clear relationships, at the landscape scale, between distinct forest types and substrate variation on 'typical' soils. Dryland landscapes of the middle Caquetá, Colombian Amazon, are covered by a complex of two intergrading tree species assemblages whose distributions are closely linked to those of two soil types (Duivenvoorden, 1995). A 480-ha watershed area in Guyana supported three distinct dryland associations of large tree species on 'typical' soils, each linked to particular soil conditions, as well as areas of heath and swamp forest (ter Steege *et al.*, 1993). This latter case illustrates the point that sophisticated analytical techniques are not needed for the development of classifications of forest types, even in the tropics: the forest types identified by ter Steege *et al.* (1993) using field inventories and multivariate statistical analysis, were the same as those previously described in much more subjective terms by Davis and Richards (in Richards, 1976) and Fanshawe (1952). Clark *et al.* (1998) found that the distributions of seven of nine tree species, over 216 ha of dryland forest in a Central American forest landscape, showed statistically significant relationships to soil type (recent alluvium, old alluvium, stream valley and residual soils). The distributions of five common palm species are also linked to substrate factors at that site (Clark *et al.*, 1995).

At the local scale, compositional variation in the vegetation is normally marked where there is marked local variation in topography and microtopography. Studies at this spatial scale have usually focused on the distribution of individual species, without delivering a verdict on whether the degree of overall variation is sufficient to warrant the naming of distinct forest types, though there is no reason why this should not be done for management purposes.

Ancient lava flows in north-eastern Costa Rica have weathered into a topography of low hills, their acid, infertile clay soils classed, in general terms, as residual (all are probably Ultisols) (Clark *et al.*, 1995; Clark *et al.*, 1998; Finegan and Camacho, 1999). Altitudinal variation over horizontal distances of 50 m may be ±20 m (B. Finegan and D. Delgado, CATIE, unpublished data). The forests are dominated, overall, by the canopy tree *P. macroloba* (Fabaceae/ Mimosoideae). The distributions of several middle and understorey palm species (Clark *et al.*, 1995; B. Finegan and D. Delgado, CATIE, unpublished data), nine canopy dicot tree species (Clark *et al.*, 1998) and individuals ≥ 2.5 cm dbh (diameter of tree at breast height, or 4.5 ft above ground level) of 32 tree and liana species (B. Finegan and D. Delgado, CATIE, unpublished data) have been analysed in relation to this topographical gradient, however, and marked variation is the norm (Table 17.1). Similarly, the distributions of a large number of woody species (all individuals > 1 cm dbh) have been analysed in a 50-ha permanent sample plot on Barro Colorado Island, Panama (Hubbell and Foster, 1986; Condit *et al.*, 1996). Simple maps showing species distributions laid over site contours often show unequivocal evidence of substrate preferences (Hubbell and Foster, 1986), while quantitative indices of preference for slope or swamp sites demonstrate that more than one-third of the > 300 species recorded in this study show a statistically significant preference for one, the other or both types of site (Condit *et al.*, 1996).

In French Guiana, soils of free or impeded drainage, with or without gleying, are distributed in a predictable way over the ridges, slopes and valleys of a series of micro-watersheds (Lescure and Boulet, 1985). Ten of 32 large tree taxa (not all identifications were made to species level) studied showed no variation of abundance in relation to soil conditions, while preference for

Table 17.1. Ecosystem diversity related to topographical variation in a managed neotropical lowland rainforest on infertile acid soils in north-eastern Costa Rica.

Size class	Hilltop	I.V.	Valley bottom	I.V.
(a) ≥ 10 cm dbh	*Pentaclethra macroloba*	14.9	*Pentaclethra macroloba*	20.3
	Ferdinandusa panamensis	6.2	*Iriartea deltoidea*	3.7
	Euterpe precatoria	3.5	*Welfia georgii*	3.4
	Tapirira guianensis	3.4	*Casearia arborea*	2.9
	Welfia georgii	3.3	*Apeiba membranacea*	2.5
	Protium ravenii	3.2	*Carapa guianensis*	2.3
	Subtotal (top six spp.)	34.5	Subtotal (top six spp.)	34.9
	Subtotal (136 other spp.)	65.5	Subtotal (145 other spp.)	65.1
	Total (142 spp.)	100	Total (151 spp.)	100
(b) 2.5–9.9 cm dbh	*Euterpe precatoria*	6.8	*Prestoea decurrens*	5.2
	Ferdinandusa panamensis	6.6	*Psychotria luxurians*	4.1
	Geonoma congesta	5.6	*Psychotria elata*	4.0
	Licaria sarapiquensis	4.1	*Warsewiczia coccinea*	3.5
	Protium pittieri	3.7	*Pentaclethra macroloba*	3.4
	Protium ravenii	3.4	*Laetia procera*	2.6
	Subtotal (top six spp.)	23.4	Subtotal (top six spp.)	22.8
	Subtotal (140 other spp.)	76.6	Subtotal (108 other spp.)	77.2
	Total (146 spp.)	100	Total (114 spp.)	100

The table gives the importance value, I.V. (calculated as relative abundance (%) + relative basal area (%) + relative frequency (%)/3), of the six most important species in two size classes, in two topographical categories. Note that while two species (the dominant canopy tree *Pentaclethra macroloba* and the palm, *Welfia georgii*) are among the most important in both topographical categories for individuals ≥10 cm dbh (diameter at breast height), the other important species are completely different on comparing hilltops with valley bottoms. The degree of variation between topographical categories appears greater in the understorey (individuals 2.5–9.9 cm dbh), where no species is among the most important in both categories. The species of highest I.V. in the understorey in each topographical category, *Euterpe precatoria* and *Prestoea decurrens*, are closely related palms. The vertical distances separating sites in the two categories are approximately 20 m. Unpublished data of B. Finegan, D. Delgado and N. Zamora.

non-gleyed, free draining soils was the most frequent response shown by the other species; a small group of species were more abundant in conditions of impeded drainage and/or gleyed soil (Lescure and Boulet, 1985). These authors emphasized the continuous spatial variation showed in this forest area, with all species present in all soil conditions, the variations found being of relative abundances and basal area.

Another reasonably well-known neotropical forest type is that dominated by *Dacryodes excelsa* and *Sloanea berteriana*, common on many islands of the

Caribbean basin but best studied on Puerto Rico, where it is known as tabonuco forest, after the local name of *D. excelsa* (Johnston, 1992). The topography of tabonuco forest in eastern Puerto Rico is extremely variable and this forest type may be thought of as consisting of two intermingled associations, one dominated by *D. excelsa* which is more abundant on ridgetops along with other species such as *Inga fagifolia*, and one dominated by *S. berteriana* which is more abundant on lower lying areas, with species such as the palm *Prestoea montana* (Basnet, 1992; Johnston, 1992). Significant compositional variation may be evident within areas as small as 0.72 ha in Puerto Rican tabonuco forest (Johnston, 1992).

The existence of ecosystem diversity in neotropical moist forests with gap disturbance regimes is much easier to demonstrate than are the factors which actually cause it. The fact that the most important and evident compositional subdivisions of forest vegetation may be made in relation to substrate water regimes has led several authors to suggest that soil water is the main factor to which plant species are responding (e.g. Lescure and Boulet, 1985; Pires and Prance, 1985). As we have seen, the distributions and abundances of species may sometimes be correlated with soil type or soil nutrient levels within dryland forest formations, however, and where species of differing nutrient requirements coexist, it is likely that ecosystem diversity is generated by a tendency for them to separate along soil fertility gradients (e.g. Herrera and Finegan, 1997). Much research remains to be done in this area, with particular emphasis on dryland forests.

3.5 Variation in forest characteristics within climatic zones: the effects of 'stand-replacing' disturbance regimes

In boreal, temperate and tropical forests, stand-replacing natural disturbances include hurricanes and lesser storms, floodplain dynamics, avalanches, landslides and mudflows, fire and (to a much lesser extent in the tropics) outbreaks of insect herbivores (White, 1979; Veblen, 1985; Clark, 1990; Whitmore, 1990; Attiwill, 1994). The reviews cited should be consulted for more detailed information on the principles of disturbance ecology as applied to forests. Here, we briefly discuss the more important stand-replacing disturbance types of neotropical moist forests, with emphasis on the generation of structurally and floristically distinct forest types, which is not usually addressed in reviews of the subject. The spatial scales considered are usually the landscape or local, though differentiation of forest characteristics at the regional level may potentially occur where stand-replacing disturbances are prevalent in one region but not in another – for example, between those parts of Central America which are frequently impacted by hurricanes and those parts which are not. In all climatic zones, post-disturbance forest communities are dominated by light-demanding tree species, which may eventually be replaced by more shade-tolerant species through succession, depending on the frequency of disturbance and the available species pool.

The effects on forests of hurricanes arising in the Caribbean Sea are relatively well studied. Most studies focus on forest recovery after single hurricanes, such as Hugo (1989; cited in Walker *et al.*, 1991) or Joan (1988; cited in Vandermeer *et al.*, 1996), though the overall characteristics of forested landscapes are undoubtedly determined by the cumulative effects of successive hurricanes. The degree of damage caused by individual hurricanes to Caribbean forests varies markedly within and between impacted areas, the extremes being mere defoliation – the most widespread damage category – and the breakage and blowdown of swathes of trees (Brokaw and Walker, 1991). Heavier damage may produce massive though localized post-hurricane regeneration of light-demanding tree species, as has been documented in Puerto Rico (Doyle, 1981) and Nicaragua (Vandermeer *et al.*, 1996). The middle-elevation forests of the north coast of Honduras have suffered severe hurricane impacts at least five times this century, the latest being those of Hurricane Fifi in 1974 and Hurricane Mitch in 1998 (Ferrando *et al.*, in preparation). Stands classified as currently mature are characterized by abundant palms and continuously regenerating canopy tree species such as *Vochysia* cf. *jefensis*, *Brosimum alicastrum* and *Calophyllum brasiliense*. Some mature forest patches, however, show an accumulation of very large (> 1 m dbh) individuals of the commercially valuable, endemic (Vásquez-G., 1994) canopy tree *Magnolia yoroconte*. Like other species of the genus (Vásquez-G., 1994), *M. yoroconte* appears to be strongly light-demanding; the patches of large *Magnolia* trees currently observed in mature stands may indicate areas blown down by a drastic 19th-century hurricane disturbance, in which the *Magnolia* was recruited, and have survived later hurricanes as the species appears to be resistent to blowdowns (cf. Boucher *et al.*, 1994). Interestingly, *M. yoroconte* is scarce in patches disturbed by Hurricane Fifi, though regeneration may be found in secondary vegetation on land abandoned by farmers during the 1970s, which in general, has a composition and structure quite different from that of hurricane-disturbed forest (Ferrando *et al.*, in preparation). These Central American forests indicate how quite complex patterns of ecosystem diversity may be imposed on a landscape by both natural and human disturbance, and how, in the case of the patches of old *M. yoroconte*, the maintenance over time of some elements of that diversity may depend on species-specific responses to hurricane disturbance.

Although most neotropical lands are outside the hurricane belt, extensive patches may be blown down in almost any moist lowland forest. Nelson *et al.* (1994) and Nelson (1994) describe fan-shaped forest blowdowns in the Brazilian Amazon, in which successional forests dominated by typical light-demanding tree taxa such as *Inga* and *Vismia* quickly regenerate. Analysis of satellite images of the whole Brazilian Amazon showed blowdowns concentrated in a band from southern Venezuela to the Brazilian state of Rondonia, where rainfall and storm frequency are greatest (Nelson *et al.*, 1994). These blowdowns may have enormous effects on local and landscape ecosystem diversity (the largest recorded was 3370 ha), but at the regional level their importance is probably limited, as detectable blowdowns never covered more than 0.21% of a given Landsat TM scene (185 × 185 km).

Of greater importance than blowdowns to the ecology of floodplain forests are the dynamics of the floodplain itself. Recent years have seen relatively intense study of this subject in western Amazon forests, with particular emphasis on south-eastern Perú (e.g. Salo *et al.*, 1986; Dumont *et al.*, 1990; Foster, 1990; Puhakka *et al.*, 1993; Terborgh *et al.*, 1996). Actively meandering rivers deposit sediments on which primary forest succession begins, while eroding existing terrain and destroying established forests. Taking together the forest area on currently active floodplains and that on old floodplains now free of the influence of rivers, the data of Salo *et al.* (1986) indicate that as much as 25% of the lowlands of western Amazonia (their study was based on the same Landsat images as were analysed by Tuomisto *et al.* (1995), as described above, and covered 500,000 km^2) may be covered by primary successional forest. Dumont *et al.* (1990) have shown, however, that areas affected by floodplain dynamics are in fact limited to well-defined depressions produced by faulting and folding of the two principal western Amazonian tectonic units. Most of the modern dryland forest of western Amazonia, they conclude, has not been affected by floodplain dynamics for at least 10,000 years. Forest ecosystem diversity is a significant feature of the large areas of currently active floodplains, being made up of a rather predictable sequence of successional communities which develops over time on the alluvial terraces (Salo *et al.*, 1986; Terborgh *et al.*, 1996). Foster (1990) estimates that due to the high rates of migration of rivers, forest age in the Manu River floodplain of Perú may rarely exceed 200 years.

3.6 Logging and floristic and structural variation in neotropical moist forests

The extraction of timber is a disturbance and as such, changes habitat conditions for forest plants and animals. Changes in habitat conditions will bring about changes in species composition of logged areas. Basic principles (Pickett and Carpenter, 1995) indicate that the degree and duration of compositional change will depend on the intensity and frequency of the disturbance caused by logging. However, within a given pre-logging forest type, logged areas will have a different species composition to unlogged areas, so that from the point of view of the present chapter, logging contributes to ecosystem diversity. From the point of view of biodiversity, it is to be expected that logging and other forest management interventions will simplify forests (Noss, 1999). Little detailed information is available on changes of plant species composition brought about by logging of neotropical forests, however, and work in tropical forests in general has focused on logging damage or on compositional and structural change in the first years after intervention, and the pioneer tree species which regenerate in disturbed areas (Johns, 1997). Again returning to basic principles, it is to be expected that logged forest will have a greater relative abundance of light-demanding tree species than unlogged and that populations of some species typical of the undisturbed forest may be reduced or perhaps lost (ITTO, 1999a). Research on longer-term changes in forest composition and on non-pioneer species is badly needed, however. While stands of pioneer trees are a visually impressive aspect of areas in which canopy opening by logging is drastic, they may last as little as 10 years (Finegan, 1996) and more lasting changes in forest composition have to be sought by studies of other species groups. Trees of canopy tree species of the long-lived pioneer guild as defined by Finegan (1996), for example, may live for more than a century, while the simplistic scenario that common and widespread colonists will replace old-growth specialists in managed forests (ITTO, 1999a) appears largely untested in the neotropics.

4 Ecosystem-level biodiversity and C&I for sustainability

4.1 Ecosystem and landscape level biodiversity conservation goals for forest management units

In this section we first set out some brief conclusions, derived from preceding discussion, which form the basis for the setting of biodiversity conservation goals for the ecosystem level in FMUs. In the following section, ecosystem-level biodiversity C&I from existing sets are reviewed and a possible generic set for FMUs is illustrated.

Ecosystem diversity is a real, important and ubiquitous facet of forest biodiversity, though its measurement, and the mapping of ecosystems (forest types), often require subjective or arbitrary decisions regarding the location of boundaries which are more than justified by the operational utility of the results. Ecosystem diversity is evident at scales of analysis which vary from the local to the regional, all of which may be relevant, in different ways, to FMUs. At the large scales which typify many FMUs, the ecosystem is more appropriate than the species as the level for the assessment and management of resources: firstly, as a measure of biodiversity, and secondly, as a basis for the planning and execution of activities related to both production and conservation. The identification and mapping of forest types, at appropriate spatial scales, should therefore be considered a basic step in the process of sustainable forest management. From the conservation standpoint, one goal of forest management should be to conserve both the individual ecosystems of the FMU and their patterns across the landscapes or parts of landscapes covered by the FMU. By making progress towards this goal, management would also make significant progress towards additional goals concerned with the conservation of biodiversity at the species level. It is obviously insufficient, however, to assess biodiversity conservation simply on the basis of the area and pattern of forest types in the FMU – ecosystems will be modified, often simplified (Noss, 1999), by management, even if their area and pattern are conserved. The areas and proportions of each ecosystem which are intervened, and the type and degree of intervention, must therefore also be assessed by C&I.

These conclusions established, the rest of this section reviews the approaches taken to ecosystem-level evaluation in the current C&I processes, and suggests future directions for research and development work.

4.2 Ecosystem-level criteria and indicators for biodiversity

Approaches taken to conservation at the ecosystem level in current C&I processes

Regarding the assessment of biodiversity conservation at the ecosystem level, common threads run across the different C&I sets from regional or national to FMU levels. To highlight this commonality, we include regional and national C&I in the following brief review. The ecosystem-level indicators proposed by the different C&I processes may be interpreted as an application of the coarse-filter/gap analysis approach, taking forest types as both units of biodiversity to be conserved in themselves, whose conservation will additionally represent a major step towards the conservation of a large proportion of species and genetic diversity.

Starting at the national level, 'Conservation of Biological Diversity' is Criterion 1 of the Montreal Process C&I, and five indicators of this criterion concern ecosystem-level biodiversity (see Appendix 4 of Lammerts van Beuren and Blom (1997) who also discuss the other C&I sets mentioned below). Among these indicators are the following, which exemplify the type of approach taken by the regional and national C&I sets:

a. 'extent of area by forest type relative to total area',
b. 'extent of area by age class or successional stage', and
c. 'extent of area by forest type in protected area categories as defined by IUCN or other classification systems'.

Criterion 4 of the Tarapoto Process national-level C&I, Conservation of Forest Cover and Biological Diversity, emphasizes the three hierarchical levels of biodiversity, though only one of eight indicators refers to forest types, this having the same wording as indicator (c) of the Montreal C&I (see above; Toledo, 1996). The regional and national C&I of the Central American (Lepaterique) process also include indicators similar to the Montreal indicator

(c) under their criteria regarding the maintenance of forest ecosystem services and biodiversity (FAO/CCAD/CCAB-AP, 1997a). Criterion 4 of the Helsinki Process C&I (these are also national-level) is Maintenance, Conservation and Appropriate Enhancement of Biological Diversity, it also emphasizes the three levels of biodiversity (Patosaari, 1997). The Helsinki Process defines concept areas in relation to each criterion, and one of those linked to Criterion 4 is Representative, Rare and Vulnerable Forest Ecosystems. The quantitative indicator defined for this Concept Area is: 'Changes in the area of natural and ancient semi-natural forest types, strictly protected forest reserves and forests protected by special management regime'.

At the FMU level, the Forest Stewardship Council Principle 6, Environmental Impact, refers to the conservation of 'unique and fragile ecosystems and landscapes' (FSC, 1996). FSC's Criterion 6.3 is that ecological functions and values (these include genetic, species and ecosystem-level diversity) shall be maintained intact, enhanced or restored; Criterion 6.4 refers to the protection in their natural state of 'representative examples of existing ecosystems'. ITTO's guidelines for biodiversity conservation in managed forests similarly recommend that undisturbed refuge areas be maintained in logged forests and should include all forest types in the locality, and highlight 'areas with unusual land forms, geology, or other physical features not adequately represented in totally protected areas (TPAs)' and 'areas of forest type not represented in TPAs' as being of special importance to conservation (ITTO, 1999a). These guidelines do not match the indicators set out under Criterion 5 of ITTO's C&I set, however, which do not refer to forest types at the FMU level (ITTO, 1999b).

Other FMU C&I sets are less explicit concerning ecosystem-level biodiversity. The explanatory text of Tarapoto's Criterion 10 for the FMU, Conservation of Forest Ecosystems, states that recognition of the ecological differences between forest types is of fundamental importance to sustainability (Toledo, 1996). No indicators related to forest types are presented, however, the focus instead being on the relative areas of production and protection forest within the FMU. The Lepaterique FMU C&I emerged as draft proposals from two workshops carried out in February 1997 (e.g. FAO/CCAD/CCAB-AP, 1997b). These proposals, like the Tarapoto set, subsume ecosystem-level diversity in indicators related to proportions of forest area allocated to production and protection forest.

The most complete set of biodiversity C&I for the FMU level is undoubtedly that of the CIFOR process. One of the basic assumptions of this process is that ecosystem structure (including taxonomic composition), function and resilience should be metrics of concern for ecosystem management (Prabhu *et al.*, 1996). The biodiversity C&I originally proposed (Prabhu *et al.*, 1996) were deemed insufficient and replaced by a much more extensive set, in which it was proposed that each indicator be evaluated by measurement of one or more verifiers, these defined as 'data or information that enhances the specificity or ease of assessment of an indicator' (Stork *et al.*, 1997). The first group of indicators and verifiers in this C&I set refers to landscape pattern and includes several verifiers related to the attributes of different vegetation types within landscapes.

It is not the purpose of the present chapter to provide a critical review of C&I sets as they currently stand. Some comments on approaches to biodiversity conservation in FMUs are relevant, however. The ecosystem (forest type) level is clearly dealt with in the FSC Principles and Criteria, the ITTO guidelines (but not their C&I) and the CIFOR process. The Tarapoto and Lepaterique processes, on the other hand, do not make explicit reference to ecosystem diversity, though they could profitably adapt requirements to establish totally protected areas within the FMU so that these areas cover each forest type present. Tarapoto and Lepaterique, however, emphasize the existence or not of measures to protect rare or endangered species as an indicator related to biological diversity. Large vertebrates, or any vertebrate subject to hunting pressure, may require special measures for their protection in FMUs. As we have seen above, however, there is good reason to believe that the rare and endangered species approach may often not be useful or practicable (especially in the tropics) and should be at least complemented by a coarse-filter/gap analysis approach (it may be that a coarse filter/gap analysis approach is not initially possible either, but in such cases, this approach should arguably be given priority in the development of FMU information bases and in future research and development initiatives – see later). Concerning this latter point, C&I sets which focus only on the area and proportion of the whole FMU which is assigned to strict protection – as in the Lepaterique draft C&I – run obvious risks, such as the hypothetical case in which the whole area of a forest type with high commercial stocking is logged over, and the proportion of the FMU under protection is made up by a forest type with low stocking. These are some of the reasons which justify the compilation by the present authors of a possible generic set of ecosystem-level C&I for biodiversity conservation in FMUs, presented in the following section.

A possible generic set of ecosystem-level indicators and verifiers for biodiversity conservation in forest management units

In this section we compile, and briefly discuss, a possible generic set of ecosystem-level 'indicators and verifiers' (I&V) for approaching the assessment of biodiversity in FMUs. The biodiversity conservation goals which may be assessed using these I&V are set out above, and the I&V could be linked to most of the criteria related to biodiversity conservation in the sets mentioned in the present chapter. It is emphasized that assessment at the ecosystem level is relevant not only to biodiversity conservation, but also to the planning and execution of forest management in general, and that these I&V should be integrated with those for other aspects of sustainability. The I&V set out

(Table 17.2) are partly a compilation and synthesis of proposals from the C&I processes which have been discussed above, as well as information in reports of field tests of C&I such as Prabhu *et al.* (1998). Others are suggested by the present authors. The exercise is intended to be illustrative, not exhaustive. As the set compiled is generic, it makes no direct reference to the scale at which forest types could or should be identified.

Forest management should be based on an understanding of how forest characteristics such as productivity, resilience (the speed with which ecosystem characteristics recover following disturbance; Pimm, 1984) and diversity vary between forest types across that part of the landscape covered by the FMU. With specific reference to biodiversity, it is axiomatic that management should seek to conserve or restore the original ecosystem biodiversity of that part of the landscape covered by the FMU (Indicator 1). As we saw earlier from the theoretical point of view, and above from the C&I point of view, the conservation of ecosystem biodiversity may be assessed on the basis of the areas and proportions of the different types of forest and other community in the FMU (Verifier 1.1) and the landscape patterns formed (Verifier 1.2). The implementation of forest restoration strategies in deforested areas is also a relevant verifier for this indicator (Verifier 1.3), though the regenerating forest is likely to maintain a structure and composition very different from that of the original forest for many decades (see earlier). As previously emphasized, not all forest types should necessarily be considered equal in biodiversity assessments, however – some are likely to be more important than others with respect to conservation objectives, and therefore should be subject to special management regimes (Indicator 2). Awareness of, and action upon, national or regional conservation objectives with respect to the protection of different forest types, as demonstrated by the management plan and its execution, could be a useful verifier in this respect (Verifier 2.1). In situations in which conservation objectives for forest types have not yet been defined at national or regional levels, a precautionary approach by the forest manager would be to take special measures for the protection of forest types of limited area within the FMU, or of unusual characteristics (Verifier 2.2).

It appears essential to assess not only the area, proportion and landscape pattern of forest types in the landscape, and the special management measures applied to those of special importance, but also to determine the degree to which each forest type is modified – probably simplified, in terms of its compositional and structural biodiversity (Noss, 1999) – by management. This assessment is important as it would provide information both on the degree to which the original characteristics of each forest type are maintained, and therefore indirectly, on the degree to which its species diversity is likely to vary. Thus Indicator 3 assesses conservation and sustainability on the basis of the proportions of each forest type which are modified by the management process, and the type and degree of modification. The natural disturbance regime is one of the most important factors in the determination of forest

Table 17.2. Examples of indicators and verifiers (I&V) that might be used for the assessment of biodiversity conservation at the ecosystem level in forest management units.

Indicators	Verifiers	Comments
1. Cover of all forest types in the forest management unit (FMU), and the spatial patterns formed by the forest types in the landscape, are conserved	1.1 Absolute and proportional areas of each community type, both forest and non-forest, in the FMU	
	1.2 Degree of fragmentation (patch structure, connectivity and edge features), by forest types	See Stork *et al*. (1997)
	1.3 Restoration strategies implemented in deforested areas	Natural secondary succession should be considered a main option
2. Forest types of special importance for biodiversity conservation are subject to special management regimes	2.1 Management plan takes into account and, where necessary, acts upon, national and regional priorities for ecosystem conservation	
	2.2 Special measures are taken for the protection of natural forest types of limited area or unusual characteristics	See International Tropical Timber Organization (ITTO, 1999a); partially covers for the possibility that Indicator 2.1 cannot be applied
3. The modification of each forest type by management does not exceed established limits	3.1 Natural disturbance regimes are not changed	Refers to the basic dichotomy – 'stand' and 'gap' disturbance regimes
	3.2 Absolute and proportional areas of each forest type intervened, to be intervened, and in unmodified reserve areas	See Stork *et al*. (1997); assesses extent of likely simplification of the vegetation within each forest type
	3.3 Proportion of intervened areas in which forest structural and floristic recovery is underway	Essential complement to 3.2
(B.) Protection measures are effectively implemented	4.1 Strictly protected areas are clearly marked on maps and in the field	

These I&V would be appropriate for assessments under criteria such as ITTO's Criterion 5, Biological Diversity, and FSC's Criteria 6.3 and 6.4.

biodiversity (see above) and its basic characteristics (gap or stand-replacing) should not normally be changed by management (Verifier 3.1). Note that this point applies not only to changes from less (gap regimes) to more frequent and severe disturbance (stand-replacing), but also to the reverse case – changes from stand-replacing to gap regimes; the well-documented effects of fire control in temperate and boreal forests are an example of this latter case (see Noss, 1999). Beyond the maintenance of disturbance regimes, it is necessary to assess the extent of forest modification/simplification on the basis of the proportions of forest area intervened, to be intervened, or to be designated as strictly protected areas (Verifier 3.2). This latter indicator could be complemented with assessments of post-intervention forest recovery (Verifier 3.3). There is considerable potential for the application of further indicators or verifiers to the assessment of the degree to which intervened areas are modified by the management process – the habitat structure verifiers set out by Stork *et al.* (1997), for example, or some of those listed under 'community-ecosystem: structure' by Noss (1990, his table 1).

4.3 Identifying and mapping ecosystems in forest management units

How might forest managers obtain or develop an information base on the forest types in an FMU? Classifications of the vegetation types of a region, some with diagnostic keys, may already exist and in some cases, regional classifications of vegetation are already the basis of biodiversity conservation measures in guidelines or plans for forest management (e.g. Woodley and Forbes, 1997; Department of Natural Resources and the Environment, DNRE, 1998). In other situations, of course – particularly in the tropics – there may be little or no documented information on forest types, or it may not be easily accessible to the forest manager. The brief comments in the rest of this section apply mainly to situations in which information is inadequate, and in which forest types may have to be identified and mapped in an empirical way using limited information; it seems clear that where information is inadequate, however, a major contribution to capacity for the sustainable management of forests and their biodiversity would be made by regional or national initiatives to develop vegetation classifications.

The characterization of natural disturbance regimes is a basic step in the development of a classification of forest types. If stand-replacing disturbances occur, a classification based on stages in the development of regrowth can be adopted or developed. The proportions of forest area found in each stage of development, and the area of forest which is maintained in a mature or over mature condition, are likely to be important indicators of biodiversity conservation in such situations (e.g. Woodley and Forbes, 1997). If gap disturbance regimes are prevalent, information on the relationship between forest characteristics and substrate conditions should be used to approach a working classification of forest types. There is no reason why such classifications should not be subjective, at least as a starting point: subjectivity does not preclude operational utility, and multivariate statistical methods of community analysis are not necessarily superior to subjective description in this latter sense (Noss, 1987).

Many possible sources of information may be available. Forest inventory data can be adapted to the development of vegetation classifications. Local people, as in the Peruvian Amazon (Encarnación, 1985) may already use their own classifications of forest communities. Similarly, both farmers and personnel involved in forest management operations may have a clear vision of the ways in which valuable tree species are distributed over the landscape, and of the environmental factors that may affect species distributions. Especially if such tree species are common or dominant, this information may in itself be sufficient for a simple classification of forest types. Information on non-timber species which are well known to local people because of value for other purposes may also be useful. In tropical regions, palms (family Arecaceae) should be especially valuable tools for the detection of vegetation responses to environment. Many ecological studies have related the distributions and abundances of palm species to local environmental variation (e.g. Clark *et al.*, 1995) and as plants which have served humans for multiple purposes over millennia (e.g. Henderson *et al.*, 1995), it is likely that in a given FMU, many palm species and their habitat preferences will be known to local people. Finally, even in the complete absence of information on the vegetation, basic information on physical environments, such as soil type and topography, should give a clear general picture of the way ecosystem diversity is likely to be distributed over the FMU. Hunter *et al.* (1988) give a precedent for this latter suggestion. They propose that, as a safeguard against the effects of climate change and the contingent nature of communities, the coarse filter approach to biodiversity conservation be based more on the distributions of physical environments than on those of vegetation types. Though their paper takes a longer-term view than appears necessary for C&I evaluations of forest management, the principle is the same.

5 General conclusions

Ecosystem biodiversity is not as widely discussed or understood as biodiversity at the species level, although the nature and characteristics of natural and semi-natural communities have been a principle object of study since the inception of ecology as a science. Biodiversity at this level is real though often different ecosystems can be mapped, and biodiversity actually measured, only if the limits which separate them are subjectively or arbitrarily drawn. As long as the subjectivity or arbitrariness of limits is not forgotten, however, practical approaches to the identification and mapping of different ecosystems at appropriate scales permit the development of planning and assessment tools of potentially enormous value. The ecosystem level is a useful general measure of biodiversity at larger spatial scales. Furthermore, information on the characteristics and distributions of different forest types is widely accepted as a basic tool for the planning and execution of both forest management and biodiversity conservation: this conjunction of applications means that the ecosystem level should also be valuable for C&I assessments of biodiversity conservation in FMUs. The importance of action at the ecosystem level to biodiversity conservation in the context of forest management is recognized in most of the sets of Principles, Standards, Criteria and Indicators and Guidelines which were reviewed for the present chapter, although some FMU C&I sets do not include indicators at this level and arguably should. One biodiversity conservation goal of forest management should be to conserve the cover of each forest type in an FMU and the patterns formed by forest types across the landscape; the indicators which may be used for assessment here, such as the area and proportion of the FMU occupied by each forest type, may not require information beyond that necessary for the planning and assessment of other aspects of the management process. The coarse filter principle is the basis of the important assumption that in conserving ecosystem biodiversity, a major contribution to the conservation of FMU biodiversity at the species level will be made, without the necessity for detailed knowledge of biodiversity at that level. Assessment of the conservation of forest types and landscape pattern must be complemented, however, by assessment of the degree to which each forest type is modified, probably simplified, by management operations. Worldwide, the degree to which an ecosystem-level approach to the management of forests and their biodiversity is, or could be, applied, varies enormously. In situations in which information is inadequate, regional or national initiatives for the development of vegetation classifications at appropriate scales could make a significant contribution to capacity to manage forests and their biodiversity sustainably.

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Note

1 In many situations there are excellent reasons for using either the term and concept 'biodiversity' as strictly defined, or the more value-neutral, academic term and concept of 'diversity' (e.g. Pielou, 1995). Both are used in the present chapter, however, as they are so intertwined with respect to our objectives.

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18 Assessing the Success of Off-reserve Forest Management in Contributing to Biodiversity Conservation

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National reserve systems of protected areas have emerged as the foundation of strategies for biodiversity conservation. Recognition that protected areas do not function as islands isolated from their broader environments emphasizes the complementary role of off-reserve management in achieving biodiversity conservation objectives. The success of off-reserve forest management cannot therefore be judged in isolation from the broader context of its joint contribution, with that of the reserve system, to the achievement of conservation objectives. This assessment is best made on a bio-regional[1] basis.

Indicators of the success of off-reserve forest management will thus differ from those of forest management within reserves in so far as the off-reserve forests play different roles in meeting bio-regional conservation objectives. Principles from which such indicators might be developed are:

- a bio-regional basis for conservation planning;
- agreement and clear articulation of conservation objectives;
- assessment of landscape-scale functionality and process;
- assessment of landscape- and stand-level recovery potential;
- agreement of the framework within which indicators are formulated; and
- the capacity for data collection by a range of interested parties.

1 Introduction

Policies and programmes for biodiversity conservation have two principal foci – the identification and establishment of a reserve system, and the management of forests outside reserves. The balance between these two interdependent means for achieving biodiversity conservation objectives is more a matter of circumstance than science, and thus varies between nations. Other than in exceptional cases, of which New Zealand might be one example, a range of economic and social – and therefore political – forces preclude the reservation of entire national forest estates from harvesting of wood and non-wood products – even if this were desirable from the perspective of biodiversity conservation, which itself remains a matter of debate.

Consequently, the management of forests and forested lands (e.g. traditional or contemporary agroecosystems – e.g. Halladay and Gilmour, 1995; Hughes, 1998) outside reserve systems, assessed in terms of their contribution to biodiversity conservation objectives, is of primary concern to the achievement of those objectives. A considerable body of previous work (e.g. Hale and Lamb, 1997; Stork *et al.*, 1997; Bachmann *et al.*, 1998; Saunders *et al.*, 1998), and other chapters in this volume, address this topic in terms of both principles and specific elements. Our purpose is not to repeat these contributions, but to consider the topic in the context of the theme of this book, namely the perspectives of stakeholders, including those of forest policy makers and managers. This rationale identifies three key elements:

- the articulation and agreement of biodiversity conservation objectives;
- acknowledgement that the success of off-reserve management in biodiversity conservation can be properly judged only in the broader context of its joint contribution with that of the reserve system;
- a focus on indicators which relate to this contextual role of off-reserve forests in contributing to the achievement of biodiversity conservation objectives.

This chapter reviews relevant contexts before considering these elements.

2 Contexts

2.1 Biodiversity conservation objectives

Biodiversity conservation objectives are usually articulated in terms of a suite of related elements, such as those developed for Australia's forests (Australian and New Zealand Environment and Conservation Council/(Australian) Ministerial Council on Forestry, Fisheries and Aquaculture (ANZECC/MCFFA) 1997):

- to maintain ecological processes and the dynamics of forest ecosystems in their landscape context;

- to maintain viable examples of forest ecosystems throughout their natural ranges;
- to maintain viable populations of native forest species throughout their natural ranges; and
- to maintain the genetic diversity of native species.

While it is recognized that there is a variety of means to achieve this end, the concept of a reserve system of protected areas – the ultimate expression and focus of *in situ* conservation – has emerged as the foundation of conservation strategies (e.g. Western and Wright, 1994; World Commission for Protected Areas/IUCN, 1997). The declaration (e.g. ANZECC/MCFFA, 1997) or promotion (e.g. Dudley *et al.*, 1996) of area targets for reserve systems exemplifies this strategy. The identification of priority areas for protection is, of course, a critical element in any strategy for the conservation of forest genetic diversity; relevant issues have recently been reviewed by, amongst others, Bachmann *et al.* (1998). An array of sophisticated approaches has been developed to aid selection of priority areas (e.g. Margules and Redhead, 1995; Pressey and Logan, 1997).

Acknowledgement that protected areas do not function as islands isolated from their broader environments has fostered the development of a parallel suite of policies which seek to reinforce the effectiveness of the reserve system through complementary off-reserve management. Our thinking about both the nature of reserve and off-reserve areas, and their roles in achieving biodiversity conservation objectives, continues to evolve – as exemplified, for example, by the reassessment of concepts of protected area function and management (e.g. Western and Wright, 1994; World Commission for Protected Areas/IUCN, 1997) or the reinterpretation of IUCN's Protected Area categories (Dudley and Stolton, 1998). The common theme is that achieving biodiversity conservation objectives requires more than the declaration of conservation reserves; the establishment of systems of protected areas is a necessary but insufficient condition for the conservation of biodiversity.

2.2 The case for off-reserve management

The history of establishment of protected areas – typically on sites less favoured for agriculture or production forestry – implies that existing national reserve systems almost invariably represent a biased sample of ecosystems and populations, with an over-representation of uplands and slopes, sites of lower fertility and stands of lesser economic value (Ledig, 1988; Kanowski *et al.*, 1997). Similarly, because few have been established or managed according to principles of population genetics, they do not necessarily comprise viable populations of forest species. Contemporary approaches to the identification and establishment of comprehensive, adequate and representative reserve systems – such as those currently underway in Australia (e.g. ANZECC/ MCFFA,

1997) – recognize and seek to address these constraints, whilst acknowledging the complementary contribution of forests outside reserves.

While ideal reserve models assist the identification and design of protected areas for biodiversity conservation, they also demonstrate the limits of the role of protected areas. The mobility of many forest animal species, the extensive geographic distribution of most tree species, the reproductive biology of tree species and the high levels of gene flow between populations, and the large areas associated with minimum viable populations of many tree and animal species, emphasize the essential contribution of forests outside reserves to the conservation of populations represented within protected areas. However well designed and well managed the protected area system, it is through the management of forests and trees outside reserves that much *in situ* conservation of forest biological diversity will be realized (Kanowski *et al*., 1997). Whilst conservation policy is now recognizing this reality and the challenges it poses (e.g. Hale and Lamb, 1997) – challenges which are particularly strong where a large proportion of off-reserve forests are in private or traditional ownership (e.g. Tasmania, Tasmanian Public Land Use Commission, 1997; Vanuatu, Tacconi and Bennett, 1997) – our management practices, on- and off-reserve, have yet to address them substantively. The clarification of objectives for off-reserve management, and the development of criteria and indicators (C&I) against which to assess policy and practice, are essential to help us move from rhetoric towards reality.

2.3 Clarifying biodiversity conservation objectives for off-reserve management

The contributions of forests outside reserves to the achievement of biodiversity conservation objectives will vary across the landscape and over time. One key biodiversity conservation objective for forests off-reserve might be to maintain, at any given time, landscape-scale ecosystem and population functionality. Because the biodiversity conservation value of any particular protected area will vary over time as forest ecosystems change as a result of both predictable (e.g. ecological succession) and unpredictable (e.g. wildfire) events, a second key biodiversity conservation objective for forests off-reserve might be to ensure that they offer insurance against the more- and less-predictable loss of biodiversity from the reserve system. The value of off-reserve forests in these terms will vary according to the impacts of nature and of their management, and with their function in the landscape, in relation to those of protected areas.

In terms of this second objective, there are – as in life more generally – varying opinions about the form this insurance should take and how comprehensive it should be. For example, some stakeholders argue that forests outside reserves should be managed to retain at pre-management levels all species and all within-species variation; others are prepared to accept temporary local diminution or extinction, subject to maintenance at a regional scale of some (unspecified) level of representation of each species and within-species variation; still others argue that off-reserve forests should be managed to conserve only a portion of regional biodiversity. Such differences of opinion reflect both variation in value judgements and scientific uncertainty, and help explain why clear and specific objectives for off-reserve management have been difficult to agree upon and to articulate.

2.4 C&I of successful off-reserve management

The status of international and complementary national processes to develop C&I against which to assess the sustainability of forest management or the state of the environment – processes of which the meeting that was a forerunner of this volume was a part – have been reviewed elsewhere (e.g. Grayson and Maynard, 1997; Wijewardana *et al.*, 1997; Dudley and Jeanrenaud, 1998). This work has led to the development of various suites of criteria and/or indicators to assess the conservation of biodiversity – for example, those focused on processes that generate and maintain biodiversity in tropical forests (Stork *et al.*, 1997), or those developed for the Australian environment by Saunders *et al.* (1998). In the discussion below, we suggest a set of principles which we believe should be embodied in indicators to assess the success of off-reserve forest management in contributing to biodiversity conservation objectives.

3 Assessing the contribution of off-reserve forest management to biodiversity conservation – some principles

The most fundamental requirement for developing indicators of sustainable management outside protected areas is the agreement and expression of clear objectives for that management. That these objectives have rarely been articulated suggests that our ways of thinking about the relationships between forest management and conservation have been inadequate; we believe biodiversity conservation objectives might best be developed from the principles discussed below. Although agreement amongst disparate stakeholder groups of the objectives based on these principles might not be straightforward, we believe it will be facilitated by a focus on these foundations.

3.1 Begin from a bio-regional basis

The case for using biogeographical regions as the basis for conservation planning has been well made elsewhere (e.g. ANZECC/MCFFA, 1997; Pressey and Logan, 1997; Dudley and Stolton, 1998; Saunders *et al.*, 1998). Regions defined on the basis of their environmental characteristics offer a biologically meaningful basis for identifying and addressing conservation goals, and thus the respective contributions of forests within and outside reserves.

3.2 Accept a continuum of contributions to conservation

The focus on protected areas as foundations for meeting biodiversity conservation objectives has tended to diminish the potential role of off-reserve forests in contributing to conservation goals. This focus seems inadvertently to perpetuate the sense of an inherent and inevitable tension between conservation and production objectives, rather than the more contemporary paradigm articulated in *Our Common Future* (WCED, 1987) and subsequently of the mutually supportive relationship between conservation and development.

The new conservation paradigm could also be articulated in terms which contrast it with the earlier development paradigm. It would acknowledge that all forests represent some level of biodiversity, and thus have the potential to contribute to conservation objectives; one can thus define a spectrum or continuum of contributions – for example, as illustrated more generally by Pressey and Logan (1997). Conceiving of conservation contributions in these terms should help overcome one of the more unfortunate legacies of the 'conservation versus production' debate: the emergence of dichotomy, rather than a synergy, between management for conservation and that for other objectives.

In terms of the achievement of biodiversity conservation objectives, a continuum concept suggests that we should allow varying levels of contribution from different elements of the landscape – from, for example, forests in protected areas and those outside reserves. The focus thus shifts to the achievement of conservation objectives on a bio-regional scale, rather than their attainment in any one subset of that landscape.

The principles of landscape ecology and of adaptive management (e.g. Margules and Lindenmayer, 1996; Kohm and Franklin, 1997; Ludwig *et al.*, 1997), which recognize the importance of the biogeographical context and the limits to knowledge, thus help define essential elements of off-reserve forest management (Kanowski *et al.*, 1997). Indicators derived from these principles might assess:

- the maintenance or restoration of connectivity between protected areas;
- the maintenance of heterogeneity across the forest landscape;
- the maintenance of structural complexity and floristic diversity within forest stands;
- the use of an array of management strategies implemented at different spatial scales; and
- the state of processes that generate and maintain genetic structure and diversity (e.g. Stork *et al.*, 1997).

Guidelines such as those developed by Lamb *et al*. (1998) for the maintenance of arboreal habitat in off-reserve forests in Queensland exemplify how such principles might be translated into terms which indicators could assess.

3.3 Facilitate the assessment of trade-offs

The issue of trade-offs – assessing the costs and benefits of a particular forest management regime in terms of conservation, economic and social outcomes – is at the heart of the debate about off-reserve management. Trade-offs are possible if we accept the principle of a continuum of contributions; they are easy to make conceptually but rather more challenging to translate into practice. The varying stand-level management regimes likely to result from different forms of trade-off are a means of delivering contrasting conservation objectives. Because many management regimes – including those which are non-interventionist – favour some elements of the biota at the expense of others (e.g. Cork and Cateling, 1997), a variety of management regimes is likely to be necessary to achieve conservation objectives at a landscape scale.

In these terms, the development of indicators which help assess the nature of trade-offs is critical to the agreement of off-reserve management regimes. Such an assessment is embodied in tools such as BioRap (Margules and Redhead, 1995) and those currently under development by Environment Australia (A. Taplin, 1998, Canberra, personal communication), in which assessed or assigned worth for a diversity of forest values is incorporated in the decision-making process.

3.4 Assess the insurance value of off-reserve forests

If we accept that one objective of the management of forests outside reserves is to provide insurance against the loss of biodiversity within the reserve system, our focus becomes the maintenance of their capability to support the full range of biodiversity if the need arises. Indicators of the extent to which this objective is satisfied are elusive, but similar to those sought in relation to ecosystem health and vitality.

One example of a theoretical framework for such indicators is the Landscape Function Analysis proposed for rangelands and forest systems by Ludwig *et al*. (1997). The strength of such a framework is that it seeks to identify the thresholds beyond which the system is incapable of returning to its original state. Other examples are the persistence of seed stores, maintenance of soil structure and fertility, and presence of sufficient connections with reserves at landscape scales to allow recolonization by plants and animals once habitat characteristics have been regenerated. Indicators of landscape functionality have been discussed by, amongst others, Margules and Lindenmayer (1996).

3.5 Agree on the framework for formulation of indicators

Saunders *et al.* (1998; following Department of Environment, Sport and Tourism, DEST, 1994) suggest a set of principles from which indicators should be developed and against which they should be judged; these are reproduced in Box 18.1. The weighting accorded each selection criterion in the box is likely to vary amongst stakeholder groups, depending upon their interests, roles and

Box 18.1. Environmental indicators – biodiversity (reproduced from Saunders *et al.*, 1998).

The key set of indicators is defined as the minimum set which, if properly monitored, provides rigorous data describing the major trends in, and impacts on, Australian biological diversity. This key set should include: indicators that describe pressures exerted on biological diversity; indicators of its condition or state; and indicators of responses to the pressures, or to changes in the condition or state. The set of indicators should be considered at three levels of biological organization – ecosystems, species and genes – and should be as comprehensive as possible without being unwieldy.

The selection criteria for national environmental indicators are listed below (from DEST, 1994); the set of key indicators should meet as many of these as possible.

Each indicator should:

1. Serve as a robust indicator of environmental change;
2. Reflect a fundamental or highly valued aspect of the environment;
3. Be either national in scope or applicable to regional environmental issues of national significance;
4. Provide an early warning of potential problems;
5. Be capable of being monitored to provide statistically verifiable and reproducible data that show trends over time and, preferably, apply to a broad range of environmental regions;
6. Be scientifically credible;
7. Be easy to understand;
8. Be monitored regularly with relative ease;
9. Be cost-effective;
10. Have relevance to policy and management needs;
11. Contribute to monitoring of progress towards implementing commitment in nationally significant environmental policies;
12. Where possible and appropriate, facilitate community involvement;
13. Contribute to the fulfilment of reporting obligations under international agreements;
14. Where possible and appropriate, use existing commercial and managerial indicators; and
15. Where possible and appropriate, be consistent and comparable with other countries' and State and Territory indicators.

responsibilities. Hypothetically and for illustrative purposes only, it might be that governments assign particular importance to criteria 3, 6, 9, 10, 11 and 15; that forest managers are more concerned with criteria 4, 7, 9, 10 and 14; that the forest-based industries focus on criteria 6, 7, 9 and 14; that environmental groups are primarily concerned with criteria 1, 2, 4, 5, 6, 11 and 12; or that the scientific community emphasizes criteria 1, 5, 6 and 11. Any set of indicators agreed for assessing off-reserve management will necessarily have to strike a balance between utility, feasibility and stakeholder preference. It should be possible for interested parties to reach consensus about appropriate indicators; relative weighting processes such as those proposed by Colfer (1995) may facilitate agreement or help in resolution of disagreements.

3.6 Broaden the monitoring network

One of the greatest constraints to the development of practical indicators is the demands they impose on data collection (e.g. Montreal Process Implementation Group, 1998). Others concerned with environmental research and monitoring have faced similar challenges and developed some innovate responses, principally by broadening the network of those associated with data collection. For example, a cross-section of the Australian community – including interest and community groups, individuals and schoolchildren – contribute voluntarily to data collection in topics as diverse as forest fauna and flora, water quality and meteorology (Alexandra *et al.*, 1996). Whilst there are limits to the role that such interested parties can assume, there is clearly potential to develop monitoring regimes which both engage and educate the participants and inform forest management. Various Australian forest management agencies, for example, have recognized these opportunities and initiated programmes which capitalize on them. Such partnerships may also be advantageous in addressing the particular challenges of monitoring forest management on private lands.

4 Conclusions

The achievement of biodiversity conservation objectives relies on the success of management both within and outside reserves. We do not believe the success of reserve or off-reserve management can be assessed until biodiversity conservation objectives have been agreed and articulated on a bio-regional basis.

Indicators of the success of off-reserve forest management will differ from those of forest management within reserves in so far as the off-reserve forests play different roles in meeting bio-regional conservation objectives. We suggest a suite of principles from which such indicators might be developed:

- a bio-regional basis for conservation planning;

- agreement and clear articulation of conservation objectives;
- assessment of landscape-scale functionality and process;
- assessment of landscape- and stand-level recovery potential;
- agreement of the framework within which indicators are formulated; and
- the capacity for data collection by a range of interested parties.

Note

1 A bio-region can be defined as 'a land and water territory whose limits are defined not by political boundaries, but by the geographical limits of human communities and ecological systems' (Bridgewater *et al.*, 1995).

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Spatial Patterns and Fragmentation: Indicators for Conserving Biodiversity in Forest Landscapes

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Despite the variety of definitions, or perhaps because of it, ecologically sustainable forest management (ESFM) is now the dominant paradigm guiding resource use in forest landscapes. Stakeholders have diverse and changing expectations from forests which need to be addressed by forest managers and planners. A toolbox of indicators is available to help assess and monitor success in meeting these expectations sustainably, ranging from attributes of landscapes or habitats to distributions and abundances of indicator species. These indicators cannot be perfect or definitive, but it is important that they lead to a shared and ecologically sound understanding of what really happens in the forest. This chapter focuses on indicators of spatial pattern and forest fragmentation, a major process impacting on biodiversity. It develops an understanding upon which key indicators can be developed, and considers alternative approaches to assessing and monitoring fragmentation and its biological effects at the landscape scale.

Forest fragmentation and its effects are a complex problem. Currently, there are few tested and proven indicators for assessing and monitoring the forest fragmentation process. Scale is critical to understanding of the problem and developing meaningful indicators. When we as humans view forests at different scales, we view spatial patterns, and may use such patterns to describe or summarize what we see. Particular patterns (e.g. patch or edge distributions) have important implications for biodiversity conservation. The few empirical studies show that relationships vary according to scale, and the spatial and temporal context and species under consideration. It is important that these relationships are studied, so that

sustainable forest management can be improved adaptively, and fragmentation indicators developed and their limitations understood. It is also clear that patterns need to be managed now at the landscape scale, with imperfect knowledge, to produce a diversity of structures and spatial patterns necessary to cope with the uncertainties in biodiversity conservation.

1 Introduction

People are an integral part of forest ecosystems. Some people live in forests, and far more visit forests for work or recreation, or use forest products to sustain their lives in more open environments. All have different needs and expectations from forests. However, increasing global demands for wood products, coupled with rapid population growth in developing countries, has resulted in fragmentation of forest landscapes and a decline in their capacity to maintain biodiversity, in temperate and tropical environments (Kohm and Franklin, 1997; Laurance and Bierregaard, 1997). The result is that most forest landscapes exist in various states of structural modification (Fig. 19.1).

ESFM requires the balancing of these increasing human pressures with the capacity of forests to produce forest products and maintain biodiversity. Currently, there are few agreed, tested or proven indicators for assessing and monitoring the sustainability of forest management practices. However, relevant and ecologically sound criteria and indicators (C&I) are increasingly necessary to guide the development and evaluation of adaptive approaches to forest management and planning in the 21st century (Kohm and Franklin, 1997). A number of international processes are endeavouring to develop such indicators (e.g. Anon., 1995; Commonwealth of Australia, 1997).

This chapter begins with some introductory comments about the purpose of indicators, establishing a few points that may be overlooked in recent literature on the subject. The chapter emphasizes the need to consider spatial pattern and context explicitly in developing indicators of ecological sustainability. Its central aim is to discuss landscape-scale patterns and processes and help

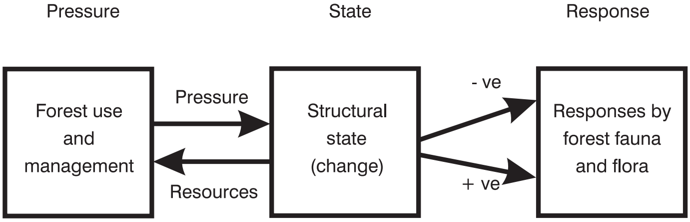

Fig. 19.1. Simple pressure–state–response model showing multiple species responses to human-induced change in forest landscapes.

formulate ecological principles for assessing and monitoring fragmentation of forest landscapes, and for understanding the varied potential responses of biological populations. Three possible approaches to assessing and monitoring fragmentation are presented, based on spatial indicators (key landscape attributes), 'robust empirical indicators' and groups of indicator species. The chapter focuses on Australian experience, and is supported by examples from Australia's sub-tropical and temperate forests and woodlands. However, it is expected that many of the principles and concepts are applicable to other tropical, sub-tropical and temperate biomes.

2 The purpose of indicators

Despite the variety of definitions, or perhaps because of it, sustainable development and management is now the dominant paradigm guiding resource use at all scales (Ferguson, 1996; Smith and McDonald, 1998). In sustainable forest management (SFM) and planning, the main purpose of indicators is to assess and monitor whether forests are being managed sustainably, and to understand forest processes and their management. It is important that we select a set of indicators for formal use that are practical to measure with available resources, and help lead to shared understanding of what really happens in the forest or give early warning of environmental change. We should not shirk the task of developing such a list, but neither should we expect it to be a definitive or comprehensive list of indicators that attracts universal acceptance. This will be an ongoing process, where new indicators are proposed and tested to assess whether societies' needs and expectations of forests are being met sustainably.

An important criterion in selecting sustainability indicators is that they should be relevant to stakeholders in forest management. Community participation is clearly needed, both in setting SFM and planning goals and in monitoring outcomes (e.g. Dargavel *et al.*, 1988; Saunders, 1990; Davie, 1997; Dorricott *et al.*, 1997). It is also essential to recognize the cultural diversity of people and their multiple uses of and expectations from forests. For example, many traditional societies such as the Penan of Borneo and Australian Aborigines live in tropical rainforests and open savannah woodlands, using them as a complete integrated habitat for food, shelter, medicine and spiritual fulfilment. Other forests and woodlands are generally used by traditional societies in combination with other habitats such as wetlands and grasslands. Often forests are used as an essential source of fuel, forage for domestic animals, meat (from hunting wild animals), food supplements (e.g. fruit and honey), clothing or adornment (skins, fur and feathers), building materials or medicinal plants. In systems of shifting agriculture, forests are used as a source of new or nutritionally restored land to be cleared for temporary use in food production. In the modern age, most of the world's people use forests to varying degrees as a supplementary habitat, living in more open environments (often created by clearing forests) and using forests as a source of water, wood, livelihood, economic wealth, recreation or spiritual refreshment. Even people who never visit a forest expect the world's forests to continue to supply these goods and services and to conserve biodiversity on a sustainable basis.

Indicators have an important role in developing ecological understanding. Forest ecosystems are far too complex for us to expect to ever understand more than a few components and their interactions. Yet ESFM demands that we manage to sustain the whole ecosystem, and monitor appropriate subjects which indicate how well we are doing. This has led to calls for 'ecosystem management' as an alternative to single-species management, though there are potential pitfalls in either approach (Simberloff, 1998). Either approach requires indicators to be monitored. There is no such thing as a perfect indicator (telling us everything we wish to know), but that should not deter us from seeking indicators that are informative, useful and efficient to monitor.

Forest management may lead to fragmentation of some habitat elements (e.g. old forest) and not others, and understanding the differential effects on species is a complex issue. Data on the distribution or abundance of selected indicators may help determine the importance of fragmentation or other consequences of management on a range of biota. Potential indicators include: elements of habitat known to be sensitive to management (e.g. old or hollow-bearing trees); metrics of spatial pattern or fragmentation (McGarigal and Marks, 1994); and plant or animal species believed to act as keystone species, umbrella species, flagship species or indicators of key ecological processes (Landres *et al.*, 1988; Noss, 1990; Paine, 1995; Power *et al.*, 1996; Simberloff, 1998). Target species or assemblages of species may respond quite differently to forest fragmentation or spatial and temporal factors acting at a range of scales (Fig. 19.1, also Faith and Walker, 1996; Niemi *et al.*, 1997; Howard *et al.*, 1998; Oliver *et al.*, 1998). Hence it is naive to expect that spatial or temporal changes in a single group will reflect general changes in biodiversity of all other groups, or that changes in abundance of one species will inform us precisely about changes in any other species. Principles of ecological isolation (Grinnell, 1917, 1924, quoted in Lack, 1971) dictate that no co-existing species will behave in exactly the same way as each other, but this does not prevent us from looking for broad patterns of general response, and learning from them if they exist. Unfortunately, such patterns have often proved elusive (Niemi *et al.*, 1997). Ideally, we should seek to develop quantitative linkages between species assemblages and forest fragmentation or management. However, less rigorous spatial or species-based indicators should not be rejected because they are imperfect, or we will never progress to operational monitoring and adaptive management. It is equally important that their limitations are recognized in making management decisions.

Some final points deserve emphasis. The purpose of indicators is that they should be easily measured and inform us about something more complex (the ecosystem), not vice versa. It is futile to measure complex biological systems merely to indicate the extent of human-induced disturbance, as the latter can usually be measured more directly. The prime purpose of measuring biological systems is to assess whether biodiversity goals are being met, and to assess the biological effects of human management or disturbance so that thresholds can be identified and management decisions made accordingly. Management should focus on goals set by the community of stakeholders, and indicators should be used to help that community determine whether those goals are being met sustainably across the landscape. Sets of indicators should include some that are not the main focus of management, or we will institutionalize too narrow a management focus.

The best value will be obtained from monitoring programmes if they are designed to answer research questions about known disturbances amenable to management (e.g. logging, fire or clearing). However, it is also important that the monitoring systems should be able to detect any incidental changes that arise from processes that have not been recognized *a priori*, as discussed under species groups (diurnal birds and ground dwelling mammals) later in this chapter. Many historical losses of vertebrate fauna from Australia and New Zealand have involved the impacts of introduced mammals in combination with habitat change (Diamond and Veitch, 1981; Burbidge and McKenzie, 1989), and some current changes appear to relate to legacies from past management rather than obvious contemporary actions (Recher and Lim, 1990; Robinson, 1993).

3 Spatial pattern, fragmentation and scale

> Forestry traditionally has concerned itself with individual stands and has been reluctant to deal with issues at larger spatial scales, even though some of these – such as the cumulative impact of fragmentation – are of overwhelming importance . . .. Spatial patterns are important and foresters and resources are at risk when they ignore this principle.
>
> (Kohm and Franklin, 1997, p. 9)

3.1 Forest landscapes and spatial pattern

Forest landscapes have natural levels of spatial and temporal heterogeneity. These spatial and temporal patterns are driven by environmental resource variability and disturbance/recovery regimes such as fire and logging. Spatial patterns may be heterogeneous or homogeneous, depending on the scale of observation. Distinctive, relatively homogeneous landscape elements or patches may be observed within a larger, more heterogeneous landscape. This hierarchy of patterns is the hallmark of landscapes (Urban *et al.*, 1987), and is a consequence of complexity within ecological communities (Szaro, 1996) and the footprint of human land use on the landscape. Human-induced disturbances tend to alter the 'natural' heterogeneity and spatial patterning of forest landscapes. (The term 'natural' is used to indicate lack of discernible impact from modern technological society.) The result is that most forest landscapes exist in various states of structural modification (Fig. 19.1). Within production forests, logging with regeneration produces a young forest, reducing and fragmenting old-growth or late-successional habitats (Loyn, 1985; Spies and Franklin, 1996). Outside production forests, clearing results in the spatial fragmentation of intact forest landscapes into increasingly isolated forest patches (Saunders *et al.*, 1987; Bennett, 1990a,b; Bennett *et al.*, 1994, 1998; Greenberg, 1996; Laurance and Bierregaard, 1997). Both processes may fragment forest habitats into small or isolated patches, with important consequences for biodiversity.

Spatial pattern is the physical layout of all patches in the landscape (Dunning *et al.*, 1992). It is one of two components of landscape structure, the other being landscape composition or the relative amounts of each habitat type contained in the landscape. Landscape structure regulates landscape function or the flow and interaction of energy, materials, and species among the component ecosystems or patches (Forman and Godron, 1986). Hence, changes in both landscape composition and spatial pattern have important implications for flows and interactions which constitute landscape function. Breakdown in landscape function can produce dysfunctional landscapes, causing vital soil nutrient and water resources to be lost from the system (Tongway and Ludwig, 1994), or biological populations to become spatially divided (Hanski and Gilpin, 1997). Fragmentation of populations is usually considered detrimental, as it may accelerate local extinctions and reduce the chance of subsequent repopulation. However, there may be benefits, as isolated populations can escape effects of broadscale stresses such as effects of introduced animals or extensive wildfire. These factors need to be considered on a case-by-case basis, and conservation strategies usually aim for a mixed approach and multiple reserves each above a minimum threshold in size.

3.2 Fragmentation and spatial pattern

In all forest biomes there are different forest types and various spatial–temporal patterns of human-induced change. Many forest landscapes are both 'naturally' fragmented and seasonally dynamic due to spatial and temporal variation in environmental resources and stresses. This provides the context upon which human-induced fragmentation and its effects on biological populations must be assessed.

Fragmentation of spatial pattern involves a disruption of continuity (Lord and Norton, 1990): the breaking up of habitats into small parcels (Forman, 1995). As with spatial pattern, it is not restricted to a particular scale, or to the spatial domain as opposed to the temporal or functional domain (Lord and Norton, 1990). It can be applied to any domain in which continuity is important in the functioning of ecosystems and landscapes. It affects both plants and animals, and the flows of water and nutrients across the landscape. However, what constitutes a fragmented forest landscape will depend on the nature of the forested landscape and the species or flows being examined (Tickle *et al.*, 1998). Both factors must be given explicit consideration in developing meaningful indicators of forest fragmentation.

There is no single fragmented landscape spatial pattern which can be identified and generically assessed in the development of fragmentation indicators (Tickle *et al.*, 1998). Fragmentation affects spatial patterns in multiple ways. It involves more than changes in the size and isolation of habitat patches. When a landscape is fragmented, habitats are replaced by other habitats, patch boundaries are often sharpened and patch context changed, and connectivity altered (Wiens, 1997). The prevailing paradigm in landscape analysis assumes a mainly dichotomous distinction between focal habitat patch and a hostile surrounding matrix (Forman and Godron, 1986). Corridors link focal habitat patches across the hostile matrix. Under this model, a fragmented landscape consists of small patches of suitable habitat embedded in a matrix in which the habitat has been destroyed. Remnant forest patches within an agricultural landscape generally could be considered a patch–matrix model. This is because most of the surrounding non-forest matrix is unsuitable for the majority of forest-dwelling species (e.g. Diamond, 1975; Howe, 1984; Forman and Godron, 1986; Loyn, 1987; Bennett, 1990a,b; Saunders, 1990; Saunders *et al.*, 1991; Barrett *et al.*, 1994; Bennett *et al.*, 1994, 1998; Catterall *et al.*, 1997). In this landscape type, the values of forest patches for fauna have been shown to be influenced by the amount of forest in the broader landscape (Bennett and Ford, 1997) and linear corridors between remnant patches (Bennett, 1990a).

McIntyre and Hobbs (1999) argue that the patch–matrix–corridor model is too simplistic to capture the range of possible landscape configurations resulting from varying intensities of human modification (Fig. 19.2). Many landscapes have more than two habitat types with varying levels of suitability. These landscapes are described as mosaic or variegated landscapes (McIntyre and Barrett, 1992; McIntyre, 1994; Wiens, 1995, 1997). Here, the matrix can perform important habitat functions and function differently from truly fragmented landscapes with a hostile matrix. The nature and quantity of habitat in the matrix can help determine the value of more extensive habitats within a variegated landscape (Laurance, 1994; Bennett and Ford, 1997). There is a need, therefore, to emphasize human impacts in production forest landscapes in the broadest sense, with fragmented and variegated or mosaic landscapes nodes on a continuum of habitat modification or loss (McIntyre and Hobbs, 1999).

In summary, forest landscapes may exist in a series of structural states ranging from uniform landscapes through variegated or mosaic landscapes to highly fragmented landscapes. What constitutes a fragmented landscape, therefore, depends on the degree of habitat alteration and on how an organism

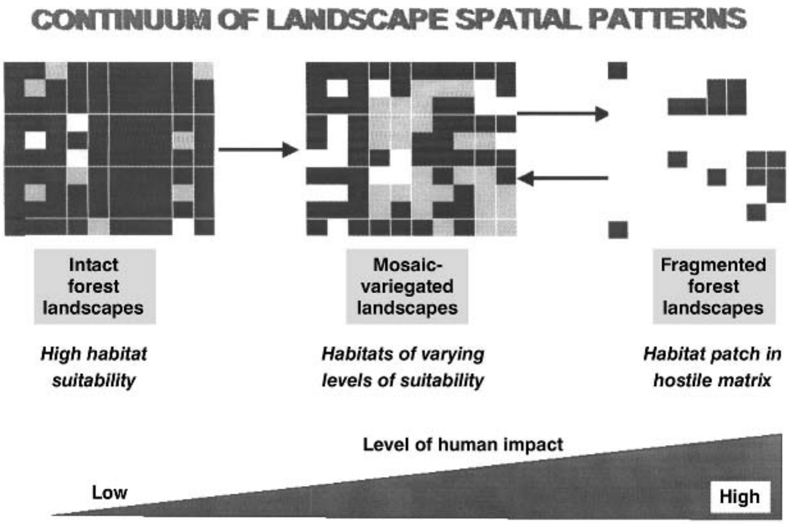

Fig. 19.2. Forest landscapes exhibit a continuum of spatial pattern ranging from intact or 'natural' landscapes to highly modified, fragmented landscapes where remnant patches of forest habitat exist with a hostile matrix. Mosaic or variegated landscapes fall along this continuum, having intermediate levels of modification. In this landscape type, the matrix is not a hostile environment, but consists of vegetation in varying states of modification, and with varying levels of habitat suitability (modified from McIntyre and Hobbs, 1999).

perceives and utilizes the spatial heterogeneity of forest landscapes. A landscape may be functionally fragmented for ants or interior-dwelling birds, and variegated for mobile bird species that make use of modified habitats in the matrix.

3.3 Organism perspective

The development of ecologically meaningful indicators of forest fragmentation requires an organism-based perspective (Pearson *et al.*, 1996). From an organism's perspective, forest landscapes are defined as dynamic mosaics of habitat resources, which occupy some spatial scale intermediate between an animal's 'normal' home range and its regional distribution (*sensu* Dunning *et al.*, 1992). This perspective is essential because the response of biological populations to changes in spatial pattern is varied (Fig. 19.1), compounded by the fact that species have different ecological tolerances to fragmentation according to their movement patterns and life history attributes (Hansen and Urban, 1992). For example, work in North America and Australia shows that bird species associated with forest interiors generally decline in abundance with fragmentation while those specializing in forest edges increase (Whitcomb *et al.*, 1981; Howe, 1984; Lynch and Wigham, 1984; Loyn, 1987; Merriam and Wegner, 1992; Barrett *et al.*, 1994; Greenberg, 1996). In North America, chipmunks made differential use of fencerow habitats according to their status as resident or transient animals (Bennett *et al.*, 1994). Resident and migratory bird species respond differently to forest fragmentation in south-east Queensland (Catterall *et al.*, 1997). The development of ecologically meaningful fragmentation indicators must take into account this multiplicity of biological responses (Tickle *et al.*, 1998). The value of indicators of fragmentation lies not in their power to describe spatial patterns, but rather their ability to describe and help understand quantitative linkages between spatial patterns at different scales and the persistence of forest fauna and fauna populations.

The concept of scale is fundamental to an organism-based perspective. When we as humans view forests in the field, or from a plane, or on a satellite image, we view spatial patterns at different scales, and may use such patterns to describe or summarize what we see. Maps represent a simplified summary of such images. Similarly, organisms perceive and respond to spatial patterns and structures in various ways, which may be quite different from a human perspective. Therefore, the importance of spatial pattern and fragmentation at different scales differs between species.

An organism perspective can be built into methodologies for developing fragmentation indicators by explicitly defining the grain (smallest patch size recognized) and ecological extent (largest scale of heterogeneity to which an organism responds) (Kotliar and Wiens, 1990). Both must be defined from an organism's perspective (McGarigal and Marks, 1994). Specification of grain and scale are essential because they determine the context of the spatial pattern and fragmentation at a particular scale for different species or species assemblages. They also provide the framework based on scale in which comparisons can be made between species.

The concept of a patch is also very much organism-dependent. Development of fragmentation indicators requires a functionally meaningful definition of patches of different levels of habitat suitability (Tickle *et al.*, 1998). Critical patch attributes include their type, size, shape and edge characteristics. In the development of fragmentation indicators, it is important that the focal patch be defined in terms of spatial and temporal patterns of resource utilization by the organism under investigation, just as patch or grain size must match the scale at which the organism operates (Wiens, 1989).

4 Indicators of forest fragmentation

4.1 Spatial indicators of fragmentation

As mentioned above, changes in spatial pattern, including habitat fragmentation, can alter the internal patterning of forest landscapes, with important consequences for the survival of biological populations. A toolbox of landscape pattern metrics is available for quantifying the spatial patterning and fragmentation of forest landscapes (e.g. O'Neill *et al.*, 1988a,b; McGarigal and Marks, 1994). A landscape metric is a spatial statistic which describes simultaneously both locational and attribute information; usually a single number in a theoretical or defined range which quantifies some aspect of the spatial distribution of the object of interest (Tickle *et al.*, 1998). A large array of metrics is available for characterizing spatial pattern and assessing fragmentation of forest landscapes. Detailed lists of landscape metrics, their interpretation and limitations, are provided by McGarigal and Marks (1994), Haines-Young and Chopping (1996) and Hargis *et al.* (1998). Those considered capable of providing information on fragmentation in forested environments can be categorized as:

1. Areal metrics – measures of landscape or patch size, interior or core area;
2. Linear metrics – measures of boundary length, width, shape at the patch level and connectivity at the landscape level; or
3. Topological metrics – measures of the spatial relationships between landscape elements in terms of dispersion, spatial association, interspersion, isolation and connectivity.

It is critical that metrics be selected and applied in a way that is ecologically meaningful to the species or species assemblages under investigation. When applied correctly, metrics serve as verifiers of change in the areal extent of habitat types and fragmentation of spatial pattern in forest landscapes.

Stork *et al.* (1997) provide a list of key verifiers to assess whether landscape pattern is maintained:

1. Area verifiers;
2. Patch structure verifiers;
3. Connectivity verifiers; and
4. Edge feature verifiers.

Landscape metrics can be selected for each verifier to assess whether critical components of landscape spatial pattern are maintained. Metrics should be sensitive to both species assemblages living in target forest landscapes and also to the structural state of the landscape (Fig. 19.2). For example, metrics serving as connectivity and edge feature verifiers account for differences in edge and matrix effects between variegated or mosaic landscapes and fragmented landscapes where the matrix is classed as hostile. In mosaic landscapes, metrics must take into account the interspersion and juxtaposition of all habitat elements in the landscape, and the juxtaposition of habitat patches of varying suitability (e.g. different ages of forest) with focal habitat patches (e.g. old-growth patches).

Area verifiers are critical for species which require large areas of forest habitat or avoid edges. The area of each habitat type is basic information for fragmentation analysis. However, it is also an important indicator of fragmentation. As the quantity of focal habitat decreases below a critical threshold for different species (With and Crist, 1995), the spatial distribution of all habitat elements (Harrison and Fahrig, 1995) as measured by patch structure, connectivity and edge feature verifiers becomes increasingly important.

Patch structure verifiers provide information on the number, size, contagion (clumpiness), dominance and fractal dimension (perimeter–area ratio) of habitat elements in the landscape. Harrison and Fahrig (1995) argue that increasing patch size, for a given amount of total habitat, increases the probability of population survival, with the positive effects of increasing patch size outweighing the associated negative effects of increasing patch isolation. Fractal dimension is a measure of shape complexity, with natural areas tending to have a more complex shape than human-altered landscapes (Krummel *et al*., 1987).

Connectivity verifiers measure the degree to which focal patches are arranged and the influence of the matrix on movement between focal patches. As a landscape becomes fragmented into smaller parcels of focal habitat, landscape connectivity may suddenly become disrupted, which may have important implications for the distribution and survival of biological populations within forest landscapes. Corridors of similar habitat linked together are thought to enhance connectivity (Forman and Godron, 1986), but dissimilar habitat elements in the matrix (e.g. mature forest patches) among which transition probabilities are high (e.g. due to low risk of predation) may also result in high connectivity. In a mosaic landscape, a patch of the same habitat type may be of different suitability, depending on the spatial heterogeneity of the surrounding landscape matrix with connectivity increasing or decreasing the likelihood of movement among specific patches. Edge verifiers are important for assessing spatial pattern for species that prefer or avoid certain types of ecotones or are susceptible to predation (Stork *et al*., 1997). However, as with connectivity, edge verifiers must be sensitive to the matrix context (Wiens, 1997) with edge contrast important in mosaic landscapes (McGarigal and Marks, 1994).

The advantage of using metrics as spatial indicators is that they can be relatively easy to apply and do not require expensive biological surveys and empirical studies. Many spatial patterns can readily be recorded by remote sensing, and temporal patterns can be determined by analysing series of remote-sensed images. Many vertebrate species respond to aspects of forest structure that can be measured remotely using airborne photography (Coops and Catling, 1997; Catling *et al*., 1998). Work is in progress to evaluate these remote-sensing methods to produce landscape-level indicators of potential habitat distribution. The main challenge is to select indicators that summarize this information in ways that are useful for managing landscapes to conserve biodiversity. Metric values are influenced by data scale (resolution or pixel size, mapping extent), number of classes and the derivation of classes. The selected mapping scale and method (classification scheme) defines patches that landscape pattern metrics are calculated upon (Tickle *et al.*, 1998). Since many forest landscapes vary continuously, classifications must be imposed on gradients of variability. Measures of spatial pattern and fragmentation, therefore, may be highly sensitive to the way in which forest landscapes have been classified.

4.2 Robust indicators of fragmentation

While landscape metrics can provide valuable synoptic information and verifiers of whether landscape spatial pattern is maintained, the identification of key landscape metrics and critical thresholds in these metrics depends on the development of robust indicators of fragmentation (Cale and Hobbs, 1994; Tickle *et al.*, 1998). Such quantitative linkages are important because critical thresholds in key metrics indicate transition ranges across which small changes in spatial pattern produce abrupt shifts in a population response. However, the development of robust empirical indicators between landscape metrics and species occurrence and decline has rarely been achieved (Cale and Hobbs, 1994; McGarigal and McComb, 1995).

Carefully designed experimental and observational studies at spatial scales relevant to species resource utilization scales are required to provide statistically robust measures of the relationship between landscape metrics and species' response to forest fragmentation and landscape change (Tickle *et al.*, 1998). The purpose of such studies should be to identify causal linkages between species abundance, distribution and diversity, and key landscape metrics. Factors influencing species distribution, diversity and abundance in forest landscapes are multi-causal, involving both direct and indirect influences (e.g. McAlpine, 1997; McAlpine *et al.*, 1999). Forest type, floristics, fire, topography and altitude, and competition between populations for often scarce habitat resources (e.g. tree hollows) are a few of the potential influences that must be considered in developing robust indicators of fragmentation. Correlative relationships do not demonstrate causality (Kenny, 1979). Given the variability and complexity created by such a suite of processes, fragmentation metrics may provide only limited power to detect and then predict long-term trends in population dynamics. In some cases, landscape metrics may be too insensitive to be effective 'early-warning' measures of population declines.

The development of robust indicators of fragmentation should aim to identify key landscape metrics and critical thresholds in habitat abundance and spatial distribution for a variety of organisms (Tickle *et al.*, 1998). For example, McAlpine (1997) found that forest cover was a significant factor influencing the abundance of eastern grey kangaroos (*Macropus giganteus*) in mosaic rangeland landscapes of Queensland, Australia. Populations of edge-dependent kangaroos were highest in mosaic landscapes with an even distribution of forest, woodland, shrub and open habitats. However, abundance declined steeply when the proportion of the landscape occupied by forest habitats dropped below 10% of the landscape. The identification of key metrics and critical thresholds depended upon separating direct and indirect influences at three spatial scales over a 4-year period (McAlpine, 1997; McAlpine *et al.*, 1999).

Long-term studies (10–30 years) may be required to establish critical thresholds in key landscape metrics (Tickle *et al.*, 1998). Such studies specifically addressing the relationship between metrics, population decline, landscape change and scale are necessary to understand the direction and magnitude of trends in the longer-term spatial and functional impacts of forest fragmentation. Retrospective studies are useful, where different landscapes have been fragmented differentially over different timescales. Sometimes such studies have provided evidence for continuing loss of species from fragmented systems over time. For example, models of bird diversity in fragmented forests of south-eastern Australia implied a continuing loss of about one species per decade from individual forest patches, in addition to the effect of patch size already recognized (Loyn, 1987). Short time-scales may show enough spatial isolation of patches to cause local extinctions in particular patches but not show results of slower processes such as inter-patch genetic isolation. That process must be examined at the longer time and broader spatial scale at which a population operates (Merriam, 1994).

5 Species groups for monitoring as indicators

In developing a programme for monitoring fragmentation and its effects, it makes little sense to focus on single species because whole suites of species can usually be monitored at one time, and each one may tell us something unique about what is happening in the forest. However, it does make sense to select groups that can be monitored by a common protocol, and to focus on particular subsets of species when attempting to interpret the results. Some possible groups are considered below, with special reference to their likely response to changing spatial patterns and fragmentation in the landscape. The main conclusions are summarized in Table 19.1.

5.1 Invertebrates as indicators?

Invertebrates contribute by far the greatest component of biodiversity in terms of numbers of animal species (Yen, 1987; Majer *et al.*, 1997; Stork *et al.*, 1997; Oliver *et al.*, 1998), and need to be addressed in any serious attempt to study and conserve biodiversity. Unfortunately, there are so many species and individuals that species-level analysis is time-consuming and expensive, and usually attempted only for selected groups (e.g. beetles, ants or butterflies). The

Table 19.1. Summary of values of animal species groups for monitoring as indicators.

Group	Why this group?	Recommendations	Selected references
Invertebrates	Greatest component of biodiversity Fundamental role in ecosystem function	Should be studied at the strategic or functional level Include at species level in selected monitoring programmes	Recher *et al.* (1996) Friend and Williams (1996) Oliver *et al.* (1998)
Frogs	Sensitive to environmental change Provide early warning of change, especially in (partly) aquatic systems	Deserve further study, especially for aquatic systems within forests	Barinaga (1990) Blaustein *et al.* (1994) deMaynadier and Hunter (1995)
Owls and arboreal mammals	Dependent on old-growth elements, e.g. hollows Some aspects of ecology are well documented Owls have large home range; high trophic level	High priority in monitoring programmes Monitor at range of spatial and temporal scales according to scale of species movement Focus mainly on arboreal mammals for quantitative analysis, and monitor distributions of both groups	Lindenmayer *et al.* (1990) Milledge *et al.* (1991) Kavanagh *et al.* (1995) Nelson *et al.* (1996) Marcot and Thomas (1997) Loyn *et al.* (2001)
Diurnal native and introduced birds	Bird species occupy broad spectrum of habitat Introduced birds often most common in disturbed habitats Conspicuous indicators	High priority in monitoring programmes Monitor a wide range of habitats Consider proportions of introduced birds as one of many potential indicators	Diamond and Veitch (1981) Recher and Lim (1990) Catterall *et al.* (1997) Loyn (1998)
Ground-dwelling native and introduced mammals	Respond to structural complexity of low vegetation and impact of introduced predators Patterns of habitat use and response differ from those of arboreal mammals	Medium to high priority in monitoring programmes Monitor at fine spatial scales	Burbidge and McKenzie (1989) Bennett (1990a,b) Lawrence (1994) Coops and Catling (1997) Catling *et al.* (1998) Kinnear *et al.* (1998)

main purpose of indicators is to help people make management decisions about whether their expectations are met sustainably, and these expectations rarely extend to conservation of invertebrates at the species level, except perhaps for charismatic groups such as butterflies. We suggest that invertebrates should be studied at a strategic level, and only included at the species level in selected monitoring programmes.

Invertebrates collectively play a fundamental role in ecosystem function, and clearly deserve further study at the functional level. With so many species replacing each other spatially across regions (Friend and Williams, 1996; Recher *et al.*, 1996; Oliver *et al.*, 1998), it seems likely that there may be a substantial degree of redundancy in function between species, though this concept is not easily tested. Invertebrates operate at all spatial scales, and may include sessile and migratory life forms at different life stages within a species (New, 1995). However, in general they respond to habitat variables at smaller spatial scales than vertebrates, reflecting their small size and microhabitat requirements. Much remains to be learned about responses of invertebrates to patch size and other aspects of forest fragmentation.

5.2 Frogs as indicators?

Global declines in certain species of frog have caused general concern, and often the causes remain unclear (Barinaga, 1990; Wyman, 1990; Richards *et al.*, 1993; Blaustein *et al.*, 1994; Lips, 1998). It seems that the group is sensitive to environmental stresses quite different from those which affect other animal groups. Forest management has been shown to affect frog populations in North America (deMaynadier and Hunter, 1995) though the group has often been neglected in studies of forest management. Hence frogs need to be monitored and studied so that causes of declines can be identified and managed. In some cases, this may help identify stresses that could affect other species groups. For example, recent research on spotted tree frogs *Litoria spenceri*, an endangered riverine frog in south-eastern Australia, has shown that the tadpoles are eaten by introduced trout *Salmo* spp. and avoided by native fish (Gillespie and Hero, 1999; Gillespie and Hines, 1999). It also appears that several native fish are confined to streams where trout are absent (Jackson, 1981). Monitoring frogs may give useful information about particular habitats such as freshwater systems within forests.

Frogs can be conspicuous and easy to monitor when calling, but calling is often highly seasonal and weather-dependent. Populations of riverine species may become functionally fragmented when rivers are affected by broadscale factors such as invasion by introduced fish in lower reaches (Gillespie and Hines, 1999). Populations of many species may be adapted to natural fragmentation of aquatic habitats within forests. Little is known about further effects of forest fragmentation.

5.3 Owls and arboreal mammals as indicators?

Indicator species or groups need to be sensitive to environmental change, and the most sensitive species may be top predators that have large home ranges and depend on complex ecosystems to sustain them (umbrella species). Large forest owls are in this category, and also have a degree of public appeal that may qualify them as flagship species (*sensu* Simberloff, 1998). In Australia, all forest owls depend mainly on tree hollows for nest sites and the larger species feed extensively on arboreal mammals which also need tree hollows (Schodde and Mason, 1984). Hence they have been subject to recent research in New South Wales (Kavanagh, 1988; Kavanagh and Bamkin, 1995; Kavanagh *et al*., 1995) and Victoria (Milledge *et al*., 1991; McNabb, 1996; McCarthy *et al*., 1999; Loyn *et al*., 2001), as well as overseas (Marcot and Thomas, 1997) to develop effective conservation strategies.

In Victoria, surveys of owls using call playback have been conducted at over 1500 sites since 1996, and logistic regression models constructed using mapped habitat data. The models were field-tested and fed back on to Geographical Information Systems to predict broader distributions and select Special Protection Zones. If these strategies prove successful, it may be concluded that other parts of the ecosystem are also conserved. To this extent, large forest owls can be useful indicators of SFM. However, it should also be noted that extensive areas of forest may not support large owls, and still need to be managed sustainably. Large owls are expensive to monitor because of their nocturnal habits and sparse distribution, making it difficult to obtain adequate data for useful statistical analysis. Arboreal mammals are commoner and relatively economical to monitor, and they are known to be sensitive to forest management (Tyndale-Biscoe and Calaby, 1975; Henry and Craig, 1984; Macfarlane, 1988; Lindenmayer *et al*., 1990; Nelson *et al*., 1996). It may be prudent to design a monitoring programme with arboreal mammals as the main subject, collecting information on forest owls in the course of fieldwork but not relying on it for statistical analysis.

5.4 Diurnal native or introduced birds as indicators?

Birds are generally conspicuous (by sound if not by sight) and active by day. They are represented by manageable numbers of species, covering a diverse array of guilds which respond to different aspects of environmental change at a range of scales. For example, changes in abundance of nectar-feeding birds or hollow-dependent birds over space or time may inform us usefully about corresponding changes in those resources (nectar or hollows) and enable us to relate those to causal factors and suggest appropriate management actions. Some Australian woodland bird species have declined recently for unknown reasons (Recher and Lim, 1990; Robinson, 1993), perhaps giving us early warning of environmental change which may be unrelated to the more obvious management actions.

The general conspicuousness of birds makes them relatively easy to monitor, compared with more cryptic vertebrate or invertebrate species. Many professional and amateur biologists have a strong interest in the group, and monitoring programmes can be devised to involve large numbers of people at relatively little cost, especially when they focus on distribution rather than abundance (e.g. Blakers *et al.*, 1984). Quantitative information may vary greatly with factors such as time of day, weather and observer skill (e.g. Bell and Ferrier, 1985; Recher, 1988; Er *et al.*, 1995) and it is rarely practical to obtain absolute measures of abundance in forest habitats. However, estimates of relative abundance between habitats may be sufficient in the context of indicators. These can usually be obtained with less effort than for cryptic species by a range of methods. Effects of observer variation can be reduced by involving more than one observer at each site (Cunningham *et al.*, 1999). Prescribed searching methods such as the timed area search are buffered to some extent against sources of random variation (Loyn, 1986; Hewish and Loyn, 1989).

Experience in many parts of Australia suggests that percentages of introduced birds (individual introduced birds as percentage of all birds observed on sample counts) may provide a useful index of gross disturbance. Values tend to be very low (< 2%) in extensive ungrazed forest and rise to much greater levels after gross disturbance such as partial clearing or grazing, or subsequent weed invasion (Loyn, 1987, 1998; Loyn and French, 1991; Catterall *et al.*, 1997). However, such disturbance can be measured directly by remote sensing, as can other forms of disturbance (e.g. logging) which do not benefit introduced species. As discussed earlier, it is futile to measure complex biological systems merely as indicators of human-induced disturbance as the latter can usually be measured more directly. However, the proportion of introduced birds is easily measured and only likely to be high where disturbance processes have had major impact on biological systems. In New Zealand, introduced birds are common (and native birds scarce) in mainland forests which look pristine but have been grossly altered by introduced mammals (Diamond and Veitch, 1981). In this case, the disturbance process is less visible and less easily measured than the indirect indicator. Similarly, in some Australian environments it is possible that high proportions of introduced birds may be the most conspicuous indicator that any of a wide range of disturbance events may have occurred at some time in the recent past.

5.5 Ground-dwelling native or introduced mammals as indicators?

Ground-dwelling native mammals may be sensitive to forest management (Dickman, 1991) and respond to changes in structure of understorey or shrub layers (Coops and Catling, 1997; Catling *et al*., 1998). They have been greatly affected by habitat loss and fragmentation (e.g. Bennett, 1990a; Deacon and McNally, 1998). Predation pressures can limit their use of habitats with sparse ground cover.

Introduced mammals have played a major destructive role in the forest ecosystems of Australia, New Zealand and oceanic islands (Burbidge and McKenzie, 1989; Towns *et al*., 1997). In southern Australia, many medium-sized (critical weight range) mammal species have become rare or extinct for a range of reasons including predation from red foxes *Vulpes vulpes* (Burbidge and McKenzie, 1989; Kinnear *et al*., 1998). The role of foxes is underlined by comparing mammal faunas of Tasmania and offshore islands (where foxes are absent and native mammals abundant) with those of mainland states (where foxes are common and native species often confined to dense vegetation such as heathlands where they can escape predation). A massive contribution to biodiversity conservation would be made if fox populations could be controlled effectively in mainland Australia. The current abundance of introduced predators may be a strong indicator that biodiversity of native mammals is not well conserved in mainland Australia.

The appropriate response to this information is to devise and implement management actions that will reduce the impact of these predators. Programmes to do this are underway in several states. Monitoring should generally focus on the native species of interest, rather than on the predators themselves. Land management practices can influence numbers of introduced predators, e.g. by giving them easy access to new habitats through track construction, or by reducing their numbers through poison-baiting programmes. However, the threat to biodiversity may apply regardless of land management practices, and the responsibility for dealing with it lies with land managers even if they did not contribute substantially to its cause.

In the context of indicators, the important point is that we need a robust monitoring system that detects biological changes and links them to causal agents even if those agents are not the main focus of land management actions. The fact that timber production reduces elements of old forest (as discussed above) behoves us to strive to understand and manage that process, and develop monitoring systems that will help us do that. But we should always be aware that ecosystems are wonderfully complex, and unanticipated processes and events are always likely to enter the stage from unexpected quarters. Our monitoring systems should be robust enough to detect such changes as they occur, so that we can respond appropriately. In the case of introduced predators, such response will have to be at the broadest landscape scale, because of the large home ranges and mobility of the animals concerned.

6 A landscape perspective on forest management

Clearing, logging and fire are the main causes of structural disturbance in the forest landscape, and all can cause varying degrees of forest fragmentation (as with recent extensive fires in Sumatra, G. Baines, University of Queensland, 1998, personal communication). Landscape patterns are a product of these disturbances, superimposed on more stable patterns determined by geology and climate. Here we focus on fire and logging because they are dynamic and potentially amenable to management. Long-term forest clearing for farmland is not a sustainable form of forest management.

Effects of disturbance need to be understood at a range of spatial scales, as illustrated for temperate Australian forests in Table 19.2 (stand level) and

Table 19.2. Habitat features of mature eucalypt forest and regrowth after logging, at scale of individual forest stands.

Mature forests	Regrowth forest
Many big old trees with hollows, deeply fissured bark and epiphytes; sparse eucalypt regeneration	Fewer big old trees; dense stands of eucalypt regeneration
Broad open spaces below canopy between widely spaced large trees	Fine mosaic of small spaces, branches and foliage below canopy
Scattered shrubs and open ground	Dense stands of shrubs and wattles; little open ground except in first few years after disturbance (when lots), and in local areas such as old log landings and along tracks
Steady supply of fallen wood of a range of size classes	Pulsed supply of fallen wood, with much small material soon after logging and less subsequently
Stands may include diverse eucalypt species (in mixed eucalypt forest) and few short-lived understorey species such as wattles, *Acacia* spp.	Stands may be dominated by short-lived species which seed prolifically, e.g. silvertop, *Eucalyptus sieberi*, and wattles, *Acacia* spp.
Vegetative resprouters may be abundant in understorey, and may be many years older than the tree canopy	Vegetative resprouters may be reduced in abundance by physical disturbance during logging
Trees of several different ages	Trees mostly of a single age (after clearfelling) or several ages (after selective logging)
Resources relatively stable over time; fluctuations due to season, weather, fire and flowering patterns	Pulses of abundance of particular resources (e.g. open ground in first few years; shrubs subsequently; peeling bark from eucalypt regrowth), and seasonal fluctuations as for mature forest

Table 19.3 (landscape level). The main tension between timber production and biodiversity conservation relates to elements of old forest which do not regenerate easily on logged areas in planned rotation times, of which hollow-bearing trees are a prime example.

There are three strategies for conserving species dependent on those elements of old forest (Loyn, 1985): retaining and regrowing adequate numbers of these elements on logged areas; extending rotations well beyond the age at which the elements have reformed; or retaining and regrowing selected stands of old forest. Forest management planning has often focused on the stand retention strategy, and debate has centred on the areas, types and spatial distributions of retained forest needed in the landscape.

In developing and applying indicators for ESFM, managers need flexibility to select a mixture of strategies to provide optimal solutions on a case-by-case basis. In some forests, it may be best to segregate production and conservation objectives spatially through a stand retention strategy. In others, it may be best to integrate them through retention of old elements on coupes. Indicators should be selected to encourage this flexibility, and focus on maintaining biodiversity in the landscape but not necessarily on every forest stand.

Table 19.3. Features of eucalypt forest landscapes with or without management for intensive timber production.

	Unmanaged forests	Managed forests
Main disturbance process	Fire (variable frequency, intensity and extent; may be managed as in timber production forests)	Logging; fire patterns usually modified to reduce frequency and extent of severe fires
Habitat patterns	Lots of mature forest especially where sheltered from severe wildfire	Less mature forest, mainly on steep slopes, gullies and areas determined by management
	Lots of uneven-aged forest, where fires promote regeneration without killing overstorey	Lots of even-aged forest on regenerated logging coupes (if clearfelling is used)
	Few large areas of young even-aged forest produced in some years	Many small areas of young regrowth produced predictably each year
	Coarse-grained mosaic of age-classes, varying over time	Fine-grained mosaic of age-classes, changing progressively but gradually over time
	Extensive areas of same-age regrowth after particular fires, with scattered older trees	Many small areas of even-aged regrowth of many different ages, with scattered older trees

6.1 What is the appropriate management scale?

The management of spatial pattern and fragmentation requires management at the landscape scale rather than at the stand or management unit scale. However, forest management units differ widely between different parts of the world, from less than a hectare in some privately owned tropical forests (I. Murthy, Indian Institute of Science, 1998, personal communication) to many thousands of hectares in sparsely populated countries such as Australia. The most pragmatic approach may be to define landscape units that are as small as practical for conserving biodiversity over time, without sacrificing flexibility to select optimum combinations of management strategies. The tension between these two demands will set upper and lower limits for landscape scales to be considered for development of biodiversity indicators. Landscape units in the order of 1000–10,000 ha may be appropriate for many Australian forests, though wildfires are sometimes much larger (> 100,000 ha) (Rawson *et al.*, 1983; Friend, 1993; Loyn, 1997; Woinarski and Recher, 1997). However, this size may not be appropriate in intensively managed forests in Asia or Europe.

6.2 Patch size and landscape context: some general management principles

Generally large patches of habitat support more species than small patches (e.g. Diamond, 1975), and this has obvious implications for conservation and development of indicators. The most serious thresholds occur where abundances of animal groups (animals per unit area) begin to decline with decreasing patch size. Different animal groups may respond to patch size in different ways and at different scales (Oliver *et al.*, 1998). Often the response to patch size relates directly to habitat features that may be influenced by physical stresses across edges, with small patches more exposed to such stresses than large patches. For example, in forests of mountain ash *Eucalyptus regnans*, an arboreal mammal species, the greater glider *Petauroides volans*, was found to be more numerous in patches of old forest far from edges of 55-year-old fire regrowth than close to edges, and this pattern correlated with the numbers of old hollow-bearing trees which had been reduced by the extensive 1939 wildfires (Nelson *et al.*, 1996). The fire boundary was diffuse and its influence had extended beyond the artificial edges we could discern in the field. This is one factor which may contribute to high abundance of arboreal mammals in large patches of old ash forest (Milledge *et al.*, 1991; Incoll *et al.*, 2001).

The importance of landscape context is illustrated by studies of birds in forest patches in south-eastern Australia. Small forest patches (< 10–20 ha) in farmland were found to support low abundances of forest birds, especially when they were heavily grazed by stock (Howe, 1984; Loyn, 1987; Barrett *et al.*, 1994; Bennett *et al.*, 1998). Often they were occupied by an aggressive native honeyeater, the noisy miner *Manorina melanocephala*, which expels other birds and compounds the effects of habitat degradation in reducing biodiversity (Loyn, 1987). Noisy miners sometimes feed in pasture and behave as edge-dependent species in similar fashion to cowbirds in North America, though their deleterious effect comes from interspecific territoriality not brood parasitism as with cowbirds. Recent removal experiments have shown that some of the biodiversity of these patches can be restored by removing noisy miners (Grey *et al.*, 1997, 1998). Any robust indicator of fragmentation would ideally need to distinguish between those forest patches which contained reasonably abundant forest birds, and those which did not. Patch size is a reasonable surrogate as a first approximation, but not the whole story.

A quite different situation arises with patches of old-growth forest of mountain ash *E. regnans*, fragmented in a matrix of younger forest (regrowth from 1939 wildfires) (Loyn, 1998). Those patches contained no farmland birds or introduced species. Forest birds were as abundant in small patches as large patches, although there were some trends for particular species and guilds (e.g. hollow-dependent treecreepers followed the pattern described for greater glider). Appropriate robust indicators of fragmentation in this context should differ markedly from those in the farmland context described above. The fragmentation process involves quite different ecological processes in the two situations, and birds respond differently to patch size.

The contrast in responses described above illustrates two points on a continuum of responses from fragmented to variegated or uniform landscapes (McIntyre and Barrett, 1992; McIntyre, 1994; Wiens, 1995, 1997). The importance of context and process in interpreting effects of fragmentation has also been recognized in landscape studies of vertebrates in northern Victoria (Bennett and Ford, 1997; Bennett *et al.*, 1998), New South Wales (Goldney and Bowie, 1990); tropical Queensland (Laurance, 1994) and elsewhere. Clearly spatial indicators must be devised to describe the full range of possibilities, and empirical research is needed to determine how biota respond to any particular situation. Genetic analysis may be useful in assessing the degree to which particular species are genetically isolated in fragmented habitats. Monitoring programmes should aim to measure biological values on and off reserves, and in various categories of modified vegetation in between. The value of monitoring programmes will be greatly enhanced if they identify factors that may contribute directly to observed responses, especially if those factors can be managed.

7 Conclusion

Key points which emerge from the preceding discussion are:

- Forest fragmentation is one of several major processes threatening biodiversity worldwide.
- Forest ecosystems are far too complex for us to expect to ever understand more than a few components and their interactions, yet ESFM demands that we manage to sustain the whole ecosystem, and monitor appropriate subjects which indicate how well we are doing. We have no choice but to use indicators.
- Spatial or species-based indicators should not be rejected because they are imperfect, or we will never progress to operational monitoring and adaptive management.
- No single spatial pattern can be identified in development of fragmentation indicators. What constitutes a fragmented landscape depends on the degree of habitat alteration and how an organism perceives and utilizes spatial and temporal heterogeneity.
- Landscape metrics can serve as spatial indicators for assessing whether critical components of landscape pattern are maintained.
- In selecting species-based indicators, it is important to consider groups of species with varying life histories and scales of movement.
- Ultimately, forest managers should aim for robust quantitative linkages between species assemblages and spatial indicators. These will help identify causal links and critical thresholds.
- In developing and applying indicators for ESFM, managers need flexibility to select a mixture of strategies to provide sustainable solutions on a case-by-case basis. Indicators should be selected to encourage this flexibility, maintaining biodiversity in the landscape but not necessarily on every forest stand.
- The value of monitoring programmes will be greatly enhanced if they identify factors that may contribute directly to observed responses, especially if those factors can be managed.

A main aim of this chapter was to discuss principles for selecting useful sets of indicators, rather than to develop a definitive list. Indeed, one of our main points is that no list should be considered definitive. Nevertheless, some conclusions can be made about the sorts of indicators likely to be most useful in the context of spatial patterns and forest fragmentation. In general, metrics of spatial pattern are likely to be more economical to measure than distributions and abundance of organisms, because the former can be assessed remotely.

The most fundamental indicator is the gross amount of habitat in the landscape, if possible compared with historical amount. The distribution of habitat is also important, and may be viewed most easily on a map or satellite image. However, this oversimplifies reality and it is also necessary to quantify the landscape context containing the remaining habitat. There is a need to define and classify the focal patch and surrounding landscape elements in a way that is relevant to the species assemblages under investigation. Landscape metrics need to be selected and interpreted in the biological context, to verify if landscape spatial pattern is maintained from the organism perspective. They should serve as area verifiers, patch structure verifiers, connectivity verifiers and edge feature verifiers. Ideally, selection of key metrics should be supported by robust empirical studies to identify critical thresholds in metric values (e.g. patch size). It must be recognized that threshold values will vary with species, context and region. When these thresholds are known, it may be possible to develop robust indicators of fragmentation that reflect the management needs of specified biota. Sustainable management should ensure that these values are maintained above critical threshold levels. Research at a range of spatial and temporal scales is needed to determine these thresholds and compare management strategies in a framework of adaptive management and continuous improvement.

The most useful species groups to monitor as indicators are those that are easily monitored and contain a range of species which provide useful information about a range of environmental stresses, especially those related to management or forest fragmentation. The most fundamental organisms in the ecosystem (microorganisms, plants and invertebrates) tend to be too diverse and numerous to be monitored easily or usefully at the species level, and not enough is known about their specific responses to disturbance. At the other end of the scale, groups of species at the top of the food-chain (e.g. large owls) tend to be too sparse to be monitored efficiently or provide information at suitable scales of management. They may warrant deliberate management as umbrella species (as is done in Victoria), and be included in a broader set of indicator groups, but additional indicator species are needed that are not management targets.

We suggest that the most useful vertebrate groups to monitor at the species level include diurnal birds, frogs, arboreal mammals and critical weight range mammals. These groups include a range of species which respond in different ways to known disturbance (e.g. logging) but are also diverse enough to give some degree of warning about unexpected changes which can be confidently expected to occur, as part of the general uncertainty involved in managing biological systems (Burgman *et al.*, 1993). We should always be alert to changes in any group of species, and assess their potential to indicate processes that may require management response.

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An Approach to Indicators for Sustainable Forest Management at the Sub-national Level in European Forestry

20

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The Office National des Forêts (ONF) is in charge of French state and community forests which cover roughly 4.4 million ha on the European continent. ONF takes on all management tasks, notably planning, work supervision in the field and sale of wood products. The forests grow in diverse ecological zones. As required by official French guidelines, all publicly owned forests are managed on a multifunctional basis, integrating ecological, economic and social aims.

As a follow-up to the Strasbourg (1990) and Helsinki (1993) conferences on the protection of European forests, criteria and indicators (C&I) for the sustainable management of forests were established at the national level by the Ministry of Agriculture. It is now desirable to prepare C&I to evaluate the results of management at the level of forest entities. Already 'Pan-European operational-level guidelines for sustainable forest management' have been published. The C&I that are being developed at the operational level will have to be: (i) relevant to the European guidelines and the existing French framework; and (ii) simple and effective. In this chapter, ONF proposes a first approach based on its experience, which is not meant to be exhaustive, and is voluntarily focused on methods.

It is proposed that these C&I be defined at the level of the small forest regions – there are 309 of them throughout France – that are mapped by the National Forest Inventory service (IFN). These regions can be considered as homogeneous for some important ecological characteristics, such as bioclimatology and geology. Their internal ecological diversity is described locally by site classification systems. These regions are used as

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units for the collection of key data on French forests and are the level at which management guidelines for both public and private forests are defined.

This chapter reviews all six criteria defined in the Helsinki process: forest structure, health and vitality, production, biological diversity, protection and socio-economic function. In each domain, the indicators that may be proposed are presented. It is concluded that the forest region is a good level at which to apply these indicators. Twenty-three indicators are selected and the availability of each in French public forests is assessed. Because of the existence of the IFN, seven indicators can be identified that will be readily available within a few years, and would apply to both public and private forests. It is stressed that two kinds of sustainable forest management indicators at the operational level will be needed: most will be applied at the national level, while the others will be region-specific. In the case of public forests, only five out of the 23 proposed indicators are immediately available, and probably more than 5 years will be needed before all the others can be obtained: the full deployment of these indicators will take time. The main questions for the scientific community relate to the concepts of soil fertility and of biodiversity. Technological progress is also much needed in the fields of aerial and satellite imagery to improve forest monitoring.

The final list of indicators at the operational level will have to be agreed upon by all stakeholders, notably the forest private owners who manage roughly two-thirds of French forests. C&I will not solve all the problems of sustainable forest management: national legislation on one hand, and working agreements between forest managers and their various partners in the field on the other, will have a very important role.

1 Introduction

Following the ministerial conference on protection of European forests in Strasbourg (1990) and the United Nations Conference on Environment and Development (UNCED) 'earth summit' in Rio de Janeiro (1992), much emphasis has been put on sustainable management of forests. The Helsinki conference (1993) produced a first set of C&I that were recommended throughout Europe (Finnish Ministry . . ., 1995). The first list of indicators of sustainable management applied to French forests was published in 1995 (Ministère de l'Agriculture . . ., 1995). These indicators are relevant only at the national level. A further step was taken with the adoption of 'Pan-European operational-level guidelines for sustainable forest management' in 1998 at Lisbon (Third Ministerial Conference . . ., 1998).

This chapter is a first attempt at defining operational-level indicators applicable to French public forests. It is based on the experience of ONF. By law, ONF is in charge of French state and community forests, which cover roughly 4.4 million ha on the European continent, i.e. roughly 31% of the total French forested area (excluding that overseas). ONF takes on all management tasks, notably planning, work supervision in the field and sale of wood products. In 1997, the Office employed 12,900 persons and had a turnover of 3.5 billion French francs (~540 million Euros) (ONF, 1998). The forests entrusted to ONF's care grow in diverse climates throughout France. They are managed on a multifunctional basis, integrating ecological, economic and social objectives (ONF, 1995b).

2 What kind of indicators?

Operational-level indicators will be quantitative tools reflecting the present condition of forest entities, and change in this condition over time. They will be used by forest managers to improve both planning and *a posteriori* control, and they will appear in reports and in public relations documents. They might also find utility in evaluating management and the managers. Forest products originating from entities where indicators are positive, and/or favourably evolving, may receive a distinctive label for market purposes.

To be widely accepted, any set of indicators will have to satisfy four conditions.

1. They will have to be consistent with the criteria that were defined, with active French participation, at the pan-European level within the 'Helsinki Process'. Six pan-European criteria for sustainable forest management (SFM), complying with the Helsinki H1 resolution, were defined in 1995. During the Lisbon conference (1998), guidelines to help participating countries define their own operational-level indicators were published.
2. They will have to be as simple and cheap to collect as possible.
3. They will have to be scientifically based.
4. Since French public and private forests share the same ecological and socio-economic environments, it is highly desirable that any scheme of indicators of sustainable management should suit both forms of ownership.

3 Choice of an appropriate scale for the operational level: the forest region ('région IFN')

In the French situation, indicators defined at the level of the smallest management unit, the forest compartment, which covers from 2 to 20 ha, would clearly be unrealistic. Individual forests also seem to be too small to form useful units: state forests have an average size of 1162 ha (although 18% of them are under 100 ha), while community forests have a mean area of 177 ha (and 57% of them are under 100 ha), and private forests have a mean area of only 3 ha (99.8% of owners possess less than 100 ha).

On the other hand, the 95 'départements' that are the main administrative divisions of French territory appear to be too large (average area ~ 580,000 ha) and are very heterogeneous from an ecological point of view.

The territorial basis of the French Forest Inventory system that was created in 1958, the 'IFN[1] region' or 'forest region', fortunately lies between these extremes. These forest regions are formally defined as 'territorial divisions where environmental factors can be evaluated to be similar in the respect of forests, and where types of forests and landscapes are comparable' (IFN, 1972). They can be considered to be relatively homogeneous as regards climate, topography and geology, and so they constitute the level at which site typologies are defined. There are 309 such regions in France (mean total area ~ 180,000 ha, minimum 12,000 ha, maximum 1,144,000 ha), containing from 100 to 920,000 ha of forests (mean 50,000 ha; median 35,000 ha) – see Fig. 20.1. Much IFN data is given at the forest region level, e.g. the area covered by different species, and the volume and production split by species. A

Fig. 20.1. The 309 'forest regions' of France.

1963 law requires that guidelines for management of private forests be set within each of these regions. Since 1986, using the framework of the same regions, ONF has developed reference documents, the so-called 'Local Management Directives' – for state forests – and 'Orientations' (in English, 'Guides') – for community forests – that guide all individual forest management plans. In the French context, the forest region thus seems to be the appropriate level at which operational-level indicators should be defined.

4 Indicators proposed

The indicators below reflect the six criteria proposed at the Third Ministerial Conference (1998) in Lisbon.

Criterion 1: Maintenance and appropriate enhancement of forest resources and their contribution to global carbon cycles

Indicator 1.1

Name	Definition and unit	Aim	Updating	Data origin	Comments
Forest cover	(Area of forests)/(total area of the region) as percentage	Estimating forest cover and its evolution over time	10–15 years[2]	IFN; aerial photograph interpretation. Availability: *E*[a]	All ownerships. Already existing information

[a]Three classes are used for data availability: *E* = existing data; *NF* = data that could be available for constructing indicators in the *near future*, i.e. ~2 years, provided that the simple treatment procedures needed are developed; *LR* = data that will be available in the *long run* only, i.e. more than 5 years.

The IFN statistical instrument is essential for private forests which are not individually mapped, and which often account for a large proportion of short-term variations in forest area. On the other hand, since public forests are well documented, their area is quite precisely known. All public forests are being mapped with a Geographic Information System (GIS). IFN detects land with actual forest cover, whereas ONF has maps of land owned either by the state or communities.

Indicator 1.2

Name	Definition and unit	Aim	Updating	Data origin	Comments
Forest compactness	For example, fraction (percentage) of forests where the mean distance to a forest edge is greater than a given value	Estimating evolution of forest fragmentation over time	10–15 years	IFN aerial photograph interpretation routinely done at every inventory cycle; then, map analysis with automated GIS tools. Availability: NF	All ownerships. Non-existing information. Could already be calculated for public forests (all GIS maps are available)

This indicator could be derived from forest maps drawn by IFN. It will be necessary to define a minimum size for the forests considered, probably depending on the region. For the IFN, the minimum size of what is considered as 'forest' is 0.05 ha with a minimal width of 15 m. Research is also needed to specify in each region the critical distance to the forest edge ensuring that all forest 'attributes' are present. Scientists could also work on practical ways to take 'connecting links' of forests, such as hedges and riparian forests (ripisylves), into account.

Indicator 1.3

Name	Definition and unit	Aim	Updating	Data origin	Comments
Management plans that are in effect	Fraction of forests with an approved management plan	Estimating the extent of management planning and activity	10 years	Public forests: ONF; private forests over 25 ha contiguous: CRPF.[3] Availability: NF	All ownerships. Information existing in two separate institutions: needs only to be centralized

Approved plans comply with national forestry regulations, ensuring that forest management is oriented toward sustainability.

Indicator 1.4

Name	Definition and unit	Aim	Updating	Data origin	Comments
Growing stock	Volume; $m^3\ ha^{-1}$	Evaluate the balance, or imbalance, of growth and harvest by assessing the development of standing stock	10–15 years	IFN. Availability: E	All ownerships. Can be assessed globally, or, preferably, by forest type (IFN definition)

The IFN forest 'types' are large entities, such as 'conifer high forest with coppice', or 'beech high forest', defined in each 'département'. Research is needed to determine for each forest type the 'normal' mean level of growing stock in the forest region. The indicator will be calculated only for those forest types that cover a sufficiently large area.

Criterion 2: Maintenance of forest ecosystem health and vitality

Indicator 2.1

Name	Definition and unit	Aim	Updating	Data origin	Comments
Fraction of indigenous or acclimatized species	Fraction of the forest area where the main species are either indigenous or acclimatized	Determine the forests that are potentially at high risk due to species that are non-indigenous and non-acclimatized	10–15 years	IFN. Availability: NF	All ownerships. Indicator based on the area covered by species at the forest region level

The list of species that can be considered as acclimatized in the forest region will have to be established and periodically updated. This indicator will not be accurate enough to detect an ill-adapted provenance belonging to a globally adapted species.

Indicator 2.2

Name	Definition and unit	Aim	Updating	Data origin	Comments
Site-adapted species	Fraction of the forest area where the main species are site-adapted	Determine the forests that are potentially at high risk due to species unsuited to soil, climatic or other conditions	10 years	Forest maps: species and site distribution. Availability: LR	Public forests: not possible before 5–10 years, when site maps will be available everywhere. In process (GIS)

Research is needed to define 'site-adapted' species in the different sites of each region. This definition should take into account possible climatic change, especially in the domain of water supply. This indicator is more accurate, but more difficult to assess, than indicator 2.1.

Indicator 2.3

Name	Definition and unit	Aim	Updating	Data origin	Comments
Forest destruction	Fraction of the forested area where the forest cover is destroyed: fire, gale, avalanche, landslide, deep frost . . .	Evaluate the forest destruction rate	1 year	Mapping from ground or air. GIS. Availability in public forests: LR; in private forests: ?	All ownerships. Concerns abiotic factors. A minimum size has to be defined. Do not include reforestation failures

Interpretation of this indicator will rely on estimation of the 'normal' incidence of destruction in the region.

Indicator 2.4

Name	Definition and unit	Aim	Updating	Data origin	Comments
Forest health status	Fraction of forest area with 'health problems': frost damage, insects, fungi, snow breaks, drought effects . . .	Monitor the health of forests	1 year	Ground or air mapping. Then GIS. Availability in public forests: LR; in private forests: ?	All ownerships. Biotic and abiotic factors

The existing European 'level 1' monitoring system, which records symptoms (crown yellowing and density) only on a very large 16×16 km grid, is not precise enough at the level of forest regions (with the exception of the very large Landes region in the south-west).

This indicator is of great importance since the vitality of French forests is often affected by biotic and abiotic factors. It is needed at high frequency, yearly being optimal. Ideally, each kind of damage should be recorded on an appropriate severity scale. Such surveys, when ground-based, are very expensive and therefore rarely done. A new approach based on remote-sensing imagery, for instance from high-resolution satellites, seems necessary. Research is needed to develop automated image interpretation techniques that are forest species specific and result in reliable data in terms of damage to tree crowns (nature and intensity). Although much needed, such methods are not likely to be available to French foresters in the short term.

Indicator 2.5

Name	Definition and unit	Aim	Updating	Data origin	Comments
Importance of forest tree decline	Wood volume sold corresponding to declining trees: m^3 ha^{-1}	An estimate of the tree mortality	Can be calculated every year	ONF, existing sales documents; mainly dead, declining and wind-thrown trees. Availability: NF	Public forests. Relevant on a 5 or 10 year basis only

Declining trees are sold, with a special note on the catalogue, at regular timber auctions organized by ONF. Trees accidentally killed can be sold individually throughout the year. This indicator is not very precise, since only adult and severely declining trees are recorded. Moreover, some dead trees are not sold. However, this indicator could be very useful when indicator 2.4 is not in operation.

Indicator 2.6

Name	Definition and unit	Aim	Updating	Data origin	Comments
Extent of artificial regeneration	(Area artificially regenerated)/ (total forested area): artificial reforestation, as a percentage	Evaluate the importance and evolution of artificial regeneration	10–15 years	IFN. Availability: E. Statistical precision at the forest region level to be checked	All ownerships

This indicator is not relevant where existing stands are of low quality. In some areas (Landes), it seems justified to be more concerned about the genetic base of the material used than about the reforestation technique employed. IFN could provide the same kind of information about afforestation.

Indicator 2.7

Name	Definition and unit	Aim	Updating	Data origin	Comments
Phytocide use	Fraction of treated areas on which non-persistent phytocides are used	Promote the use of environmentally friendly products	1 year	ONF. Availability: NF	Public forests. This indicator might be obtained from the ONF information system with a more precise recording of phytocides used

Indicator 2.8

Name	Definition and unit	Aim	Updating	Data origin	Comments
Pesticide use	Ratio of areas treated with biological control to the total area where pest control is applied	Promote biological control of pests	1 year	ONF. Availability: NF	See 2.7

Criterion 3: Maintenance and encouragement of productive functions of forests (wood and non-wood)

Indicator 3.1

Name	Definition and unit	Aim	Updating	Data origin	Comments
Harvesting level	Actual felling rate compared with management plan provisions	Ensure that wood harvesting is conducted at a sustainable rate	Thinnings: 5 years; clear cuts: 10 years	ONF. Availability: NF	Public forests. Concerns only regular fellings

This indicator will have to be used jointly with indicators 1.4 and 2.5.

Indicator 3.2

Name	Definition and unit	Aim	Updating	Data origin	Comments
Damage to standing trees due to harvesting	Average value of penalties inflicted on felling and logging operators. In French francs ha^{-1}	Assess the quality of wood harvesting operations	1 year	ONF. Availability: E in state forests, NF in community forests	Public forests (enforcement of forestry code). See ONF (1995a)

The data exist for community forests and could be easily extracted from ONF files. This indicator is not general, since penalties are inflicted only when trees over 7.5 cm dbh (diameter at breast height, or 4.5ft above ground) are damaged. This indicator could not be used in Alsace and Lorraine, where ONF fells the trees and sells them at the roadside.

Indicator 3.3

Name	Definition and unit	Aim	Updating	Data origin	Comments
Road infrastructure of productive forests	Fraction of productive forest area with no adequate timber extraction infrastructure	Adequacy of forest infrastructure, and its evolution	10–15 years	IFN level: the 'département'; the possibility of getting it at forest region level must be checked. Availability: NF	All ownerships. Indicator limited to forests with a production function

Criterion 4: Maintenance, conservation and appropriate enhancement of biological diversity in forest ecosystems

Indicator 4.1

Name	Definition and unit	Aim	Updating	Data origin	Comments
Forest tree diversity	Fraction of stands with *N* or more species in the dominant part of the stand	Evaluate the diversity of dominant tree species	10–15 years	IFN. Availability: E. Statistical precision has to be checked	All ownerships. This indicator can be calculated either globally for the forest region or at the forest type (within region) level

N will have to be determined in each region. To be considered as present in the dominant part of a stand, a species will have to represent a minimum fraction, to be fixed, of the total number of trees or of the stand basal area. A Shannon index, based on dominant forest tree species frequencies (number of trees, understorey excluded), might be more informative than this rather crude indicator.

Indicator 4.2

Name	Definition and unit	Aim	Updating	Data origin	Comments
Diversity of forest structures	Shannon index based on forest structure frequencies (fraction of forest area with a given structure)	Evaluating the diversity of forest structures	10 years	ONF; data not available in all regions; implies that a unified classification for forest structure exists for all forest regions. Availability: LR	Public forests. The information will become available during the next 10 years as all forests are mapped with GIS

In the case of high forest, horizontal diversity is created by the existence of various age classes: each age class (~20 years) should thus be considered as a type. In the case of very irregular stands, such as coppice with standards or mountain conifer forests, the number of forest types may be very high: groupings might then be necessary.

Indicator 4.3

Name	Definition and unit	Aim	Updating	Data origin	Comments
Areas where specific silvicultural measures are taken to enhance biodiversity	Fraction of total forest area where specific silvicultural measures are taken: groups of ageing trees, dead trees kept standing, specific stand structure maintained . . .	Evaluating the area affected by specific biodiversity measures	10 years	ONF. The information will become available as these measures are enforced (see ONF, 1993) and as all forests are mapped with GIS. Availability: LR	Public forests. 10 years needed before this indicator is relevant

Research is very necessary to identify the need for and techniques of biodiversity enhancement. This research will help give answers to such practical questions as how many dead trees, and how many clumps of ageing trees, are needed per unit area (e.g. per 100 ha)?

Indicator 4.4

Name	Definition and unit	Aim	Updating	Data origin	Comments
Legally protected areas	Fraction of total area where a legal protection scheme applies: parks, reserves, European habitats, *in situ* genetic conservation units . . .	Evaluating the extent of protected areas	10 years	ONF. The information will become available as all forests are mapped with GIS. Availability: NF	Public forests

Indicator 4.5

Name	Definition and unit	Aim	Updating	Data origin	Comments
Control of deer (*Cervidae*) populations	Fraction of shooting objectives realized	Estimate the efficiency of game population control through hunting	1 year	Ministry of Agriculture departmental offices. Presently, shooting objectives are not established at the forest region level. Availability: NF	All ownerships

Populations of *Cervidae* are rapidly increasing all over France to such an extent that forest regeneration is locally compromised. The quality of this indicator will improve as shooting objectives are set up on the basis of more precise data: estimates of population numbers and reproduction, degree of pressure on forest vegetation . . .. In regions where hunting regulations apply to different species (red and roe deer, wild boar . . .) one indicator should be used per species.

Criterion 5: Maintenance and appropriate enhancement of protective functions in forest management (notably soil and water)

Indicator 5.1

Name	Definition and unit	Aim	Updating	Data origin	Comments
Appropriateness of forest structure to the protective function	Where the protective function is stated as important, fraction of the forests with an appropriate stand structure	Evaluate the proportion of protection forests that effectively play their role	5 years?	ONF: available on state forests in the mountains. The mapping of protection forests with GIS is under way. Availability: LR	Public forests

In the mountains, the state bought, and reforested, ~628,000 ha of protection forests at the end of the 19th century. Ten years ago, these forests were surveyed to assess the regeneration needed (ONF, 1990). The same methodology could be used to determine in what proportion of state and community forests special silvicultural treatments are needed to maintain the protective function; this survey would take 3–5 years. In the mountains, most protection forests belong either to the state or to local communities: this indicator would not involve private owners in most cases. Away from the mountains, only small patches of forests play a significant protection role: there, the ideal stand structure will have to be determined on a case-by-case basis, thus enabling the calculation of this indicator.

Indicator 5.2

Name	Definition and unit	Aim	Updating	Data origin	Comments
Ratio of water control protection work	Fraction of the protective work achieved compared with what is considered necessary	Determine the level of equipment in protection forests	10 years	ONF. Availability: LR	Public forests

This indicator deals only with water erosion since it is a general danger, whereas block felling and avalanches happen in limited areas. The level of equipment required will have to be determined by experts in the field.

Criterion 6: Maintenance of other socio-economic functions and conditions

Indicator 6.1

Name	Definition and unit	Aim	Updating	Data origin	Comments
Management of sites of specific interest to the public	Ratio of sites identified as being of interest to the public that are adequately managed or protected	Estimate the proportion of forest sites of interest where social objectives are actively met	5 years	ONF: could be drawn from management planning maps. In each forest, sites of specific interest are being surveyed. Availability: LR	Public forests

A precise definition of what adequate management means is needed. In particular, practical threshold values should be defined to help detect areas suffering from overuse by the public.

5 Conclusion

We get back to the series of four conditions given at the beginning of this chapter:

1. The forest region seems to be the appropriate unit at which to define operational-level indicators. With a pragmatic approach, taking advantage of existing knowledge and know-how, it is possible to propose an initial list of 23 indicators applicable to French public forests and fitting the 'Helsinki–Lisbon' guidelines.
2. In such a varied country as France, operational-level indicators of SFM have to cover a wide range of situations. It may therefore be necessary to divide indicators into two categories: those that will be applied in all regions, and those that will be specific to only some regions (e.g. mountains, the Mediterranean zone, highly populated areas). Thus any system of indicators will have to be flexible. Moreover, to be manageable, the total number of indicators in any region will probably have to be kept under 25–30.

Seven out of the 23 indicators proposed will be simply and cheaply available through the IFN, and will apply to both private and public forests. However, the other indicators will require specific procedures to be set up to collect and process data: in the public forests, out of 23 indicators, 18 are not available today, and of these, eight will not be available for at least 5 years. All sets of indicators will probably have to evolve over time in order to take into account the evolution of social demands on forests. The deployment of indicators will inevitably be an incremental process.

Only a minority of the proposed indicators (indicators 2.3, 4.5 and 6.1) are related to situations subject to rapid change and which therefore will have to be assessed at intervals of less than 10 years. From a practical point of view, it seems appropriate to suggest that most of the indicators be calculated on the same cycle as the IFN inventory, that is 12–15 years (ideally 10 years).

3. This first list of indicators is not satisfying from a scientist's point of view. Some indicators required in the 'Helsinki–Lisbon' documents, and proposed here, might not be relevant. For instance, assessing indicator 2.6 'Extent of artificial regeneration' implies that natural regeneration is always preferable, which is not the case when existing stands are of low quality or have a narrow genetic base. This list of indicators utilizes readily available information about familiar traits such as forest stand composition and production. In other domains, on the contrary, appropriate indicators will not be set up until research provides essential basic knowledge. This is particularly the case in the area of functional ecology of forest ecosystems. A non-exhaustive list of tasks that can be proposed to the scientific community is the following:

- assessing the carbon fixation function by investigating the functioning of stands and forest soils (notably organic layers);
- evaluating mineral budgets in forest soils, taking into account all outputs and inputs including those from bedrock and the atmosphere: the long-term balance of these budgets is critical for sustainable management;
- developing tools to better tackle biodiversity: identifying 'key' species or groups of species from a functional standpoint, developing appropriate diversity indices, and defining appropriate scales at which conservation policies have to be enforced.

Finally, as was stated under indicator 2.4, technological progress is greatly needed in the application of aerial and satellite imagery to the monitoring of forests at the forest region scale.

4. The number of new procedures that will have to be established to collect data will be greater in community than in state forests, and probably even higher in private than in public forest. Only nine of the 23 indicators will be readily available in both private and public forests; some of the remaining ones might never be accessible in private forests. Difficult questions linked to financing and coordination of data collection and processing will arise. Time will be necessary to decide which indicators are really needed, and to organize their cheap and reliable acquisition. Indicators of SFM will not be enforced jointly for public and private owners without long preparation.

Let us stress a final point. No set of indicators, whatever its quality, will ever solve all the problems of SFM. Good legislation on one hand, and a continuous dialogue in the field between the forest manager and all other stakeholders (timber companies, hikers' clubs, protection associations . . .) on the other, will remain critical for good forest management.

Notes

1 Inventaire Forestier National: National Forest Inventory, a public institution under the supervision of the Ministry of Agriculture.
2 The time interval between two IFN surveys in the same 'département' (the aim is 10 years).
3 'Centre Régional de la Propriété Forestière', regional board of private forest ownership. There are 17 CRPFs that formally approve private forest management plans and give technical support to private forest owners.

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Conclusions 21

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Criteria and indicators (C&I) are a relatively new tool that have been developed to help better define sustainable forest management (SFM), and to assist with measuring change in forest condition and output of goods and services from forests. Initial emphasis has been on development of C&I for application at the national level, but there is widespread recognition that these need to be adapted to finer scales where forest management decisions are made and implemented.

There are widely varying views amongst stakeholders on the role and value of C&I. The views range from considerable enthusiasm by policy makers, to caution and concern by scientists and forest managers, to scepticism by many conservationists. Ongoing engagement of stakeholders is important for development of a shared position on how C&I can be effectively applied. A sharing of the responsibility and cost of implementing C&I between governments and forest managers is important to future progress.

Application of C&I in forests has the following potential benefits:

- raising awareness of, and political commitment for, SFM;
- providing a tool for reporting, at a range of levels, on the state and trend in condition of forests;
- when forming part of an environmental management system, providing a way of assessing progress against management objectives, and thus supporting adaptive forest management; and
- providing an important plank for the certification of forests as sustainably managed, and the associated 'green' labelling of forest products.

Realization of several of these benefits depends upon the effective application of C&I at the forest management unit (FMU) level, and an associated commitment (including resourcing) from forest managers. The definition of FMU is in itself an issue that remains partly unresolved. What constitutes an FMU is likely to vary significantly between countries, and even within state or national boundaries (e.g. Roman-Amat *et al.*, Chapter 20, this volume). In a sense, a definition is not important, provided that stakeholders can reach consensus on what is appropriate to their local circumstances, and that C&I are being applied in a way that will provide information that can be used to improve forest management. Broad stakeholder input and development of shared approaches is critical.

There has been a range of international processes aimed at developing C&I that cover the full range of forest values (see Castaneda, Chapter 9, this volume). There is a broad consistency between the criteria used, and general agreement that they are sufficiently comprehensive to encapsulate all important forest values. The challenge is clearly to develop useful indicators for those criteria that apply to highly diverse forests and socio-economic contexts. A further critical issue is to identify interim standards or performance measures that can be used to evaluate trends in data. This is a much-neglected aspect of the practical application of C&I (Raison and Rab, Chapter 14, this volume). R&D has a critical role in providing the basis for evaluating the significance of trends in indicators. The weightings given to the various sustainability criteria will vary markedly with local circumstances, and again stakeholder engagement to work towards developing these is essential. These aspects can be effectively dealt with by embedding C&I within an environmental management system (Raison and Rab, Chapter 14, this volume).

It is important to stress that C&I are only one of several tools that can help support SFM. As emphasized in Chapter 2, participatory planning and evaluation processes are required to develop shared goals and agreed actions following the assessment of trends in data. Demonstration forests that examine management options, that include supporting R&D, and which effectively engage stakeholders, are an excellent way of advancing SFM at the 'local' scale.

Equitable access to forest information by stakeholders has been a controversial issue during the 1990s. The need for improved access to such information has been recognized in many important international fora. In 1992, the United Nations Conference on Environment and Development (UNCED) encouraged countries to explore ways for sharing information and forest databases. Five years later at a 1997 meeting of the Intergovernmental Panel on Forests (IPF), it was stressed that attention should be given to worldwide access to information systems that would encourage effective implementation of national forest programmes, and lead to improved cooperation. There is little doubt that denied or restricted access by stakeholders to forest inventory and other data for publicly owned forests has inhibited the implementation of SFM in many countries.

The social sciences are still poorly applied in forestry and this must be rectified if better indicators of social criteria are to be developed. There is a need to draw on experiences outside traditional forestry. Likewise, there is growing acceptance of the need to draw on the accumulated knowledge of indigenous peoples when addressing social and cultural aspects of SFM.

There are many challenges to realizing the potential benefits of the application of C&I in forests. The practical application of C&I is still very much in its infancy, and realistic expectations must be maintained. There has been a tendency for the expectations of policy makers to exceed the capacity for effective implementation of C&I. The expectations of all stakeholders need to be regularly tested against the capacity to support cost-effective application of C&I. This will provide transparency in relation to the costs and benefits of the use of C&I, and help to ensure that resources are invested to strengthen weak links in C&I systems. Stakeholders must judge collectively whether benefits justify the cost of applying C&I in particular circumstances.

It is important for scientists to engage in efforts to apply existing information to define indicators and develop monitoring and evaluation systems. This will involve integration of quantitative and qualitative information and the making of 'expert' judgements. Much can be learned from trialing of 'best-bet' systems that can form a basis for evolutionary improvement. There is likely to be value in commencing work with a relatively small 'core' set of indicators that is agreed amongst stakeholders.

The following is a summary of important steps in the application of C&I:

- ensure stakeholders have access to relevant forest data sets and other information;
- involve stakeholders in the definition of forest management goals and associated targets, and the specification of possible indicators, monitoring and evaluation processes;
- evaluate proposed indicators, monitoring and evaluation procedures in terms of scientific underpinning, ability to detect important change, and practicability;
- refine and agree approach between stakeholders;
- apply indicators, perhaps in case studies, and review utility for tracking agreed change in forest condition and outputs of goods and services; communicate findings to interested parties;
- modify systems in light of experience and new information.

If these steps are well linked, there is the opportunity for progressive refinement of the use of C&I to improve forest policy and to support SFM.

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1 Entries concluding with a numeral (e.g. 'Agenda 21') are enclosed in quotation marks to avoid confusion between that numeral and page numbers