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MAIN THEME: FOREST BIOLOGICAL DIVERSITY

Review of the status and trends of, and major threats to, the forest biological diversity, prepared by the Ad Hoc Technical Expert Group on Forest Biological Diversity

Note by the Executive Secretary

1. The Executive Secretary is circulating herewith, for the information of participants in the seventh meeting of the Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA), a preliminary version of the review of the status and trends of, and major threats to, the forest biological diversity, which was prepared by the Ad Hoc Technical Expert Group on Forest Biological Diversity established by the Conference of the Parties to the Convention on Biological Diversity in its decision V/4. The review is expected to be finalized for submission to the Conference of the Parties at its sixth meeting.

2. The review is being distributed in the form and language in which it was received by the Convention Secretariat.

* UNEP/CBD/SBSTTA/7/1.

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REVIEW OF THE STATUS AND TRENDS OF, AND MAJOR THREATS TO, THE FOREST BIOLOGICAL DIVERSITY

Ad hoc Technical Expert Group on Forest Biological Diversity

Preface

In its decision V/4, the Conference of the Parties decided, at its fifth meeting, held in Nairobi in May 2000, decided to establish an Ad Hoc Technical Expert Group on Forest Biological Diversity to assist the Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA) in its work on forest biological diversity. The terms of reference of the Expert Group, as contained in the annex to that decision, request the Group to:

(a)Provide advice on scientific programmes and international cooperation in research and development related to conservation and sustainable use of forest biological diversity in the context of the programme of work for forest biological diversity;

(b)Carry out a review of available information on the status and trends of, and major threats to, forest biological diversity, to identify significant gaps in that information;

(c)Identify options and suggest priority actions, time frames and relevant actors for the conservation and sustainable use of forest biological diversity for their implementation through relevant activities;

(d)Identify innovative, efficient and state-of-the-art technologies and know-how relating to assessment, planning, valuation, conservation and sustainable use of forest biological diversity and provide advice on ways and means of promoting the development and transfer of such technologies.

The Group held two meetings with the objective to produce a report fulfilling the mandate provided by SBSTTA. The first meeting took place in Montreal, Canada, from 27 November to 1 December 2000, with financial support from the Government of Canada. It elected Dr. Ian Thompson (Canada) and Mr. Gordon Patterson (United Kingdom), as Co-Chairs of the Group, and Dr N. Manokaran (Malaysia) as the Rapporteur. Its second meeting took place in Edinburgh, United Kingdom, from 23 to 27 April 2001, with financial support from the Government of the United Kingdom.

Membership of the group was from eighteen countries, as well as from NGOs and IGOs. All members participated in providing ideas and discussions towards developing a report from the Group. At the Montreal meeting, the Group progressed through a discussion paper provided by the SCBD on forest biodiversity and then divided into working groups along various themes to consider most urgent needs and recommendations. At the outset, all members of the Group agreed that loss of forest biodiversity, especially in the tropical forest biome, has reached a crisis stage that must be immediately globally addressed to arrest and avert continued broad losses. At the second meeting in Edinburgh, the working groups continued their tasks and were successful in developing recommendations in several important disciplines with respect to maintaining forest biodiversity. The underlying approach taken by the Group was to identify major problems, which result in the loss of forest biodiversity, and then to provide recommended objectives to help to resolve those problems.

The Group produced two main documents. The recent review summarises important scientific and monitoring assessments of forest biodiversity, including broad-scale information about forest loss and studies on the effects of loss on ecosystem function. The Group also took a broad approach to the problem by assessing the underlying causes of change and the effects on local communities, including indigenous peoples. The second main document, Report of the *Ad Hoc* Technical Expert Group on forest Biological Diversity (UNEP/CBD/SBSTTA/7/6), is a summary paper that also includes the main recommendations of the group. Matrices used in defining options and priority actions for conservation and sustainable use of forest biological diversity are distributed as an information document (UNEP/CBD/SBSTTA/7/INF/4).

The final version of this document was developed at a meeting of a 'writing committee' which met at the CBD offices, in Montreal in June, and which continued to finalise the documents by electronic correspondence, and with commentary from the rest of the Expert group during June and July, 2001. Thus the documents are the culmination of work by not only the AHTEG, but from comments and reviews by other experts from throughout the world.

August 1st, 2001
Ian D. Thompson .... Gordon Patterson

Sault Ste. Marie, Ontario ....Edinburgh

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Members of the Ad Hoc Technical Expert Group on Forest Biological Diversity represented eighteen countries and a wide expertise related to various aspects of forest biological diversity: Adalberto Verossimo (Brazil), Jacques Mbandji (Cameroon), Nsangou Mama (Cameroon), Ian D. Thompson (Canada), Carlos Le Quesne-Geier (Chile), Ma Keping (China), Modesto Fernandez Diaz-Silveira (Cuba), Mart Külvik (Estonia), Stefan Leiner (European Commission), A.A. Oteng Yeboah (Ghana), Shobna Nath Rai (India), Kiyoshi Nakasima (Japan), Takayuki Kawahara (Japan), N. Manokaran (Malaysia), Bakary Toure (Mali), Cecilia Nieto de Pascual Pola (Mexico), Maria C. Raposo Pereira (Mozambique), Kazimierz Rykowski (Poland), Andrey N. Filipchuk (Russian Federation) and Gordon Patterson (United Kingdom). I am very grateful for their successful work. Especially I would like to thank two co-chairs of the Group, Ian D. Thompson and Gordon Patterson for the their valuable efforts.

Many expert from major agencies and institutes, intergovernmental and non-governmental organizations participated in the work of the Group and contributed to its results: Robert Nasi (CIFOR), Pierre Sigaud (FAO), Douglas Williamson (FAO), Mario Ramos (GEF), Kanta Kumari (GEF), Gudrun Henne (Greenpeace), John Leigh (ITTO), David Hinchley (IUCN), Max Ooft (Organization of Indigenous Peoples in Suriname, OIS), Victoria Tauli-Corpuz (Tebtebbe Foundation), Gemma Smith (UNEP-WCMC), Jaime Hurtubia (UNFF), Carole Saint-Laurent (WWF) and Kathleen McKinnon (World Bank), as well as observers Ian Plesnik (Chair of the SBSTTA), Gijs van Tol (The Netherlands), Adrian Wells (United Kingdom). Michael Garforth was the facilitator for the Edinburgh meeting.

The draft of this report was reviewed externally through the Peer-Review Process. It was also posted on the web site of the Convention for the commenting by the scientific community at large. I express my gratitude for their review to following invited reviewers: Ron Ayling, Alexander V. Pugatchevsky, Perry S. Ong, Anoja Wickramasinghe. Valuable reviews and comments were given by The Ministry of the Environment in Poland, the National Focal Points of Canada and New Zealand collecting comments by many experts and scientists from their countries, Friends of the Earth International, World Rainforest Movement, International Research Institute for Maori and Indigenous Studies, Forest Peoples Programme, Gesellschaft für bedrohte Völker (Society for threatened Peoples), WWF, FERN, UNEP-WCMC, the Liaison Unit Vienna, PEFC Council Secretariat, Confederation of European Paper Industries. I wish to express my sincere gratitude to all these for their valuable contributions.

Hamdallah Zedan
Executive Secretary

EXECUTIVE SUMMARY: STATUS AND TRENDS OF FOREST BIOLOGICAL DIVERSITY AND MAJOR GAPS IN INFORMATION

I. Status and trends of forest biological diversity

Forest biological diversity should be quantified and described on a multiplicity of scales, from large forest landscapes of several thousand square kilometres, to the genetic level within individual organisms. The present report refers to forest landscapes, ecosystems, species, and genes, and considers the diversity of structure, function, and composition existing at each level. Scale is also considered in a second sense, including global, regional and local (or national), required to report activities and outcomes that address the issue of maintenance of biological diversity in forests.

Determining the current global status of forest biological diversity is somewhat problematic because of difficulties in quantifying biological diversity in a meaningful fashion. Describing biological diversity on the local or national scale for most countries may not be entirely possible, and even in countries that attempt to report on biological diversity, data on indicators are usually not well developed. Further, the extent and rate of change of the world's forests are still unclear, especially at the national level, and long-term trends are distorted by the lack of solid baseline data and inconsistent use of terms. Where forest inventory data do exist, in both the developed and developing world, the information is often outdated, of poor quality and is especially difficult to compare among regions because the data sources, as well as the definitions of forests and forest types, differ.

The present report uses the Food and Agriculture Organization of the United Nations (FAO) definition of forests, which has been set for the monitoring of global changes in forest cover and allows comparison between countries. Although there is not complete global agreement with the FAO definition of "forest", based on those FAO data 3,869 million ha of global forest remain in 2000, but there has been decline in the forest area by ca. 9.4 million ha (0.22 per cent) annually since 1990, of which most was natural forest in the tropics. Preliminary estimates show that net deforestation rates have slightly increased in tropical Africa, remained constant in Central America, and declined slightly in tropical Asia and South America. The establishment of plantation forests and reforestation activities in temperate and boreal forests of some industrialized countries have increased and led to a decline in deforestation rates in those biomes. In the tropical biome, the rate of plantation establishment has increased dramatically during the last decade. However, the Group noted that plantation forestry cannot fully compensate for deforestation of primary forest in terms of biological diversity, especially in the tropics or in temperate regions, where exotic, rapidly growing tree species have most often replaced the original stands. FAO's assessments do not encompass forest quality aspects (e.g. no clear distinction between primary and secondary forests, nor among different types of plantations), making an assessment of the quality of global forests difficult.

Box 1

Possible definition of "forest ecosystem" and "forest biological diversity" proposed by the Group

Forest ecosystem: A forest ecosystem is a dynamic complex of plant, animal and micro-organism communities, and their abiotic environment, interacting as a functional unit, where the presence of trees is essential. Humans, with their cultural, economic, and environmental needs, are an integral part of many forest ecosystems.

Forest biological diversity: Forest biological diversity means the variability among forest living organisms and the ecological processes of which they are part; this includes diversity in forests within species, between species and of ecosystems

At the broadest level, forests need to be better categorized to enable a proper global assessment of change in forest biological diversity. At the very least, it is important to distinguish between primary forests, that have not been directly influenced by humans and thus have most of their original biological diversity, and various types of secondary forests, which have regenerated following cutting or clearing and may support only a portion of the original biological diversity. Plantations are best described as a class of secondary forests, where often the major objective is wood production, although many countries are also using plantation forestry to try to recover previously degraded woodlands. Agroforests should also be considered as a distinct class of forests because, while supporting a portion of local biological diversity, they lack full species complements.

Care must be taken in reporting forest cover, relative to biological diversity, by distinguishing among these broad classes of forests, because biological diversity differs in each. There is a need to harmonize forest reporting on the national, regional, and global scales to improve understanding of forest quality change, and also to include within these reports aspects relevant to assessing biological diversity. A key enabling feature required for reporting is the use of comparable forest classification systems that can be aggregated to higher scales, from local or national scales, and that will accurately correlate to changes in forest biological diversity. Essential improvements in collecting and reporting forest data would be, for example, to distinguish between various numerical classes of canopy cover by forest type, and between primary forests, secondary forests, plantation forests and preferably, also between young forests and older forests.

On very large scales, there is clear evidence that forest biological diversity is related to total forest area, and small forest fragments retain only a small portion of the normal species complement. Globally, many primary forests have become degraded or deforested, so it is clear that forest biological diversity is rapidly declining, especially in the tropics. The capability of forests to maintain biological diversity has changed over large areas, as primary forests have been deforested or replaced by secondary forests of various qualities as a result of activities such as cutting, land-clearing, deliberate forest fires, fragmentation caused by forest road networks and conversion to agricultural lands, and the homogenization of forest stands. Far fewer intact larger blocks of primary forests now occur, compared to earlier, in all forest biomes.

Generally, species richness increases with decreasing latitude, with the highest levels of endemism in the tropics for flora and fauna. Unfortunately, knowledge and documentation of species follow the opposite trend, and many tropical species and processes remain unidentified. An important difference between tropical forests and temperate or boreal forests is the high local richness per unit of area (alpha diversity) in tropical forests and the high endemism, compared to lower alpha diversity in the other two biomes at the stand level. Temperate and boreal forests tend to have greater landscape diversity than tropical forests. Yet in all forest biomes there are areas with very high local diversity, and forest sites with high primary productivity maintain greater diversity than those with low primary productivity. These facts have important implications, which differ among the biomes, for landscape management strategies, including protected area placement and research needs for forests.

The number of threatened and endangered forest species seems to correlate with the size and quality of forest habitats, temporal and spatial continuity in the forest landscape, and with the history of forest use. The current extinction rate is far higher (1,000 to 10 000 times) than the rate at which species evolve and is at a historically high level. The majority of animal and plant species that are becoming extinct come from forest ecosystems. Current estimated rates of extinction for most higher life-forms in tropical rainforests are 1-10 per cent of those species in the next 25 years. The main direct causes of extinctions are habitat loss, due to land conversion and fragmentation of habitats, alien species invasions, and over-harvesting of forest resources, including logging. In future, climate change may be a further major factor, interacting with existing problems and contributing to extinctions (see sub-section D, below ‘Causes of forest biological diversity loss”).

The number of endangered species, as well as local extinctions of rare species, can be expected to rise because of the time delays (”extinction debt”) associated with fragmentation effects, forest loss, and declines in habitat quality. In particular, species requiring specific habitats that may be limiting, or which have large home-ranges, will become increasingly endangered. Some well-known species, such as great apes and large carnivores, are expected to become extinct due to habitat loss, over-exploitation, genetic effects of small populations and illegal hunting, in spite of the general positive attitude towards their conservation and the considerable conservation efforts.

While there is information on the genetic diversity of a few animal species and important trees, in general few such data exist. However, it is evident, that genetic diversity will be severely eroded due to forest decline (e.g. local extinctions of small, often unique populations) and that the effects of forest fragmentation and deforestation on genetic diversity have been overlooked.

Protected forest areas have increased in recent years, both in number and in area. However, globally forests are neither well protected, nor well represented in protected areas, with less than 8 per cent of the world's forest afforded some kind of protected status. Furthermore, particularly in tropical areas, only a minor portion of all the so-called protected area is actually secure. Most protected areas are small and insufficient to serve as source populations for large vertebrate species; nor do they fully protect regional species or local genetic diversity. The lack of small-scale forest classifications for all countries precludes an assessment of the representativeness of forest types in protected areas. Nevertheless, biological diversity will never be maintained by a network of protected areas alone, and sustainable management of large associated areas will also be required. Protected areas must be considered as part of a continuum of managed areas, from primary protected forest to fibre plantations.

Regardless of forest type, various characteristics develop or accumulate with forest age. Different animal and non-woody plant species associate with various stages of forest development because of these features, and so forest communities change over time in the same location. Old forests are an important category of forests, because certain species are optimally or solely associated with such forests. Key indicators for old forests are known for boreal zones and, to a lesser extent, in temperate forests, but are poorly known in tropical forests.

A body of scientific theory helps to understand biological diversity, but much remains to be understood. In particular, while biodiversity is clearly related to forest goods and services, the exact mechanistic relationships are not well understood. Further, little testing of indicators has been carried out in terms of their capability as predictors of broader changes in biological diversity, or defining the concept of forest quality and how well it can be predicted by indicators. Finally, there is a clear need to understand critical thresholds of forest change that will produce substantial losses in biological diversity, particularly among key or keystone species.

One source of information that has largely been overlooked is the traditional knowledge of indigenous peoples. Indigenous peoples have knowledge that has developed over many generations, but this knowledge has not yet been fully understood nor recognized, because the origin, nature, ways of use and transfer of this knowledge are different from Western "formal" science and scientific practices. In addition, there is often little mutual trust for sharing traditional knowledge, due to the absence of recognition of indigenous peoples and their rights.

Total forest cover is a coarse predictor of biological diversity, and much better indicators are needed to properly report status and trends of biological diversity on scales ranging from national to global. Most local forest inventories are conducted to monitor harvestable volumes, rather than to monitor biological diversity. Monitoring of biological diversity and of changes caused by forestry practices is important in order to assess the effectiveness of management and cumulative change through forest use. Adaptive management, based on consistent monitoring and comparison of biological diversity between primary and secondary forests, is an important part of the ecosystem management protocol. Surrogates for high levels of biological diversity, such as umbrella species, indicator species, key habitats and structural indicators, may help to assess and predict the effectiveness of conservation and forest management programmes. Such surrogates must be carefully selected, based on sound scientific understanding of their properties. Data on rare and threatened forest species alone are insufficient to provide a reliable picture of broader trends in biological diversity. Species that are naturally rare, or have declining populations, represent a special case for knowledge needs and management. Such species must be identified and understood in terms of the processes which affect their populations. Often, national biological diversity databases do not exist and the availability of long-term benchmark data for trends in possible indicators is rare.

Aside from the lack of useful indicators, the incomplete and non-standard forest classifications, and the need for improved science, many countries lack the necessary infrastructure to report on biological diversity. An important prerequisite in assessing the status of biological diversity is technology transfer to developing nations, along with equipment and training in the methods required to evaluate biological diversity and natural resources, and to map their distribution.

II. Ecosystem functioning and services

Forest ecosystems provide a wide array of goods and services, ranging from marketable commodities such as timber and some non-timber forest resources, to many goods and vital services that do not usually have a market value, including, for example, global climate regulation and watershed protection. These non-market goods and services are important to humans in general on the local, national, regional, and global scales and can often be essential to maintain the way of life of indigenous and local communities.

Box 2

The ecosystem approach

The implementation of the ecosystem approach to forest biological diversity, based on the description and the operational guidance endorsed by the Conference of the Parties in decision V/6, and on the principles recommended for application under the same decision, should substantially help to maintain these non-market goods and services. The ecosystem approach could be considered as a strategy for the integrated management of forests that promotes their conservation and sustainable use in an equitable way. Humans, with their cultural diversity, are an integral component of forest ecosystems. The ecosystem approach requires adaptive management to deal with the complex and dynamic nature of forest ecosystems and the absence of complete knowledge or understanding of their functioning.

According to the ecosystem approach, forest ecosystems should be managed for their intrinsic values and for the tangible benefits they provide to human beings, in a fair and equitable way. Forest ecosystem managers should consider the effects – actual or potential – of their activities on forest ecosystems, to avoid unknown or unpredictable effects on their functioning and, therefore, on their values.

Forest ecosystems should also be understood and managed in an economic context. In particular, costs and benefits in forest ecosystems should be internalized to the extent feasible. In addition, market distortions that adversely affect forest biological diversity should be reduced and incentives that promote forest biodiversity and sustainable should be aligned.

Finally, the ecosystem approach stresses that forest ecosystems should be managed within the limits of their functioning. Therefore, the conservation of their structure and functioning should be a priority target. This is a prerequisite for keeping their full values, including the goods and services forests deliver, generally for free, to human beings.

Sound forest ecosystem functioning and, therefore, the related forest goods and services depend on the maintenance of a whole range of interactions between biotic and abiotic components. Biologically diverse forests are generally thought to be more resilient and less liable to major outbreaks of pests and diseases. However, understanding the role of biological diversity in the functioning of ecosystems is a relatively new field of research and the linkages between degree of loss of forest biological diversity and the ability of forests to sustain their range of goods and services is not well known. In view of the pending large-scale extinctions, there is an urgent need to improve our understanding in this field. Identification of critical thresholds of impacts for sustaining biological diversity and other goods and services would be valuable in developing management strategies.

The dependence of indigenous peoples and local communities on forest ecosystems and biological diversity (and the resulting goods and services) is greater than in the general population. Alterations and loss of ecosystems and forests therefore have a direct negative impact on the survival of indigenous peoples and cultures. Another problem occurs in areas where there is a change in land use of previously forested areas, often triggered by the induced dependence upon non-traditional goods and adaptation to a monetary economy, instead of traditional subsistence methods. In most instances, these changes are accompanied by equal opportunities for sustainable development for indigenous peoples.

Each of the three main forest biomes (boreal, temperate, tropical) has characteristic ecological functions, and human impacts may thus have different consequences, both currently and in a historical sense.

The boreal forest biome is characterized by low species richness and by extreme contrasts in the functional attributes of important species for ecosystem processes. So the loss of a key species may have significant impacts on an ecosystem. Large-scale human activities, such as extensive logging and those that cause global climate change, may have a dramatic impact on overall ecosystem functioning and forest goods and services. Boreal forests represent 49 per cent of the total vegetation and soil carbon contained in the three biomes, and so play a key role in global climate regulation.

Biological diversity in temperate forests is determined primarily by human-induced changes in land use and forestry practices, as well as by the site quality. Greatest diversity is reached in undisturbed, natural forests and on sites with highest fertility. Human-induced land conversion, fragmentation and air pollution can cause loss of biological diversity and ecosystem function, and climate change is likely to interact with these factors to cause further unpredictable changes. Many temperate forests, especially in Europe, have been fragmented, exploited and managed for many centuries as part of cultural landscapes, and some form of continued management will be required to maintain characteristic biological diversity and a desirable range of ecosystem goods and services. Some forest types and habitats have been particularly heavily destroyed or degraded (e.g. riparian forests). Only a very small amount of primary temperate forests remains and old growth characteristics and structures, such as dead wood, are usually under-represented in the majority of secondary and plantation forests. Overall, the temperate biome is currently an important terrestrial net carbon sink.

The main characteristics of tropical forest ecosystems are their high biological richness and high endemism and, unlike boreal forest ecosystems, the number of species greatly exceeds the number of key ecological processes. This situation gives the ecosystem an apparent stability. Tropical forests are also characterized by the very slow pace of their development, which presents a difficulty in studying their ecological processes. The consequences of species loss through human activity may be delayed and even possibly offset by redundancies in functional relationships. Tropical forest soils are vulnerable to rapid degradation and erosion after logging/forest clearing, because almost all organic matter is maintained in the vegetation. Unsustainable use in tropical forests results in the loss of key animal species, which act as vectors for reproduction and dispersal of forest trees, and loss of key structural attributes such as lianas and epiphytes, which could have long-lasting effects on biological diversity and its related goods and services. But there is also some evidence that secondary tropical forests can be carefully managed to sustain traditional products as well as some of the biological diversity and other environmental services found in primary forest. Tropical forests hold 37 per cent of forest carbon but, because of deforestation and land use changes, the tropical forest biome is currently a net source of carbon dioxide to the atmosphere.

Restoration of forest biological diversity in degraded forests and deforested lands is an issue of growing importance in both the developed countries and the developing world. Most studies of forest biological diversity have focused on natural forests. However, in the future there is a need to focus more on the potential, on the regional and landscape scales, for synergy from combining management of different forests, including primary, secondary, agro-forests and forest plantations, to achieve a specified range of goods and services. Where plantation forests are created on former agricultural lands, rather than as a direct replacement for natural forest, it has the potential to restore at least some of lost forest biological diversity and other goods and services, especially where native species of local origin are used. Plantations can also help reduce the pressure on natural forests for exploitation for fuel wood and timber. The rate of restoration of forest biological diversity in different situations is poorly understood and research should be increased in this area. Although considerable diversity may develop in a few decades, full restoration to levels of forest biological diversity approaching those found in primary forests may take centuries.

III. Valuation of forest products and ecosystem services

There is a spatial and temporal mismatch between those who bear the costs of deforestation, forest change, and biological diversity loss, and those who receive the benefits. Much of this mismatch is related to lack of value attached to forest goods and services and the higher priority that is given to short-term benefits, as opposed to long-term sustainable returns from forests.

Forest values include:

(a)*Direct use values*: values arising from consumptive and non-consumptive uses of the forest, e.g. timber and fuel, extraction of genetic material, tourism.

(b)*Indirect use values*: values arising from various forest services, such as protection of watersheds and the storage of carbon.

(c)*Option values*: values reflecting a willingness to pay to conserve the option of making use of the forest, even though no current use is made of it.

(d)*Future option values*: values of learning about future benefits that would be precluded by loss of forest resources (e.g., values related to the existence of chemical active principles not discovered yet).

(e)*Non-use values* (also known as existence or passive use values): these values reflect a willingness to pay for the forest in a conserved or sustainable use state. However, the willingness to pay is unrelated to current or planned use of the forest.

(f)...........*Intrinsic values* such as moral or ethical value, spiritual, religious and cultural value.

A focus on those values that can be quantified in economic terms can be justified by the fact that forest conservation ultimately has to compete with alternative uses of forest land. Whereas the latter have reasonably clear and identifiable market values, many forest values are currently non-marketed. In a market-oriented world, therefore, forest conservation, which provides few immediate economic benefits, can easily lose out to the market values of alternative land uses, such as agriculture or plantations, unless goods and services are valued in these analyses or long-term conservation becomes more attractive because of positive incentives.

Stakeholder analysis shows that indigenous and local communities are likely to be the major losers from the conversion of forested land to other uses. They could, however, be beneficiaries of processes designed to capture values in markets, although there are serious reservations about whether introducing real markets should be contemplated for indigenous communities, where the introduction of the market economy without appropriate adaptation measures might threaten their way of life. In many parts of the world, the issue of forest use is also very much related to discussions about the rights over land, forested areas and natural resources. Thorough stakeholder analysis at all levels, from local to global, would be a valuable basis for ensuring that the interest and potential contributions of different key groups and organizations are appropriately and fully taken into account.

Sustainable forest management is, in the short term, generally less profitable in monetary terms than ecologically unsustainable forest practices, so that, in order to be favoured in the market, non-timber benefits from sustainable forests must exceed this loss of profit. Analysis of economic values of forest goods and services, including timber, fuelwood, non-timber forest resources, genetic information, recreation and amenity, watershed protection, climate buffering and non-use values suggests, first, that the dominant values are carbon storage and timber. Second, these values are not additive, since carbon is lost through logging. Third, conventional (unsustainable) logging is more profitable than sustainable timber management. Fourth, other values do not compete with carbon and timber, unless the forests have some unique features or are subject to potentially heavy demand due to proximity to towns. Unique forests (either unique in themselves or as habitat for unique species) have high values. Forests near towns have high values because of recreational possibilities and the use of non-timber forest products and fuelwood. Fifth, non-use values for "general" forests are very modest.

There is an urgent need for more research to validate these conclusions and also to establish the direct economic value of biological diversity other than for genetic information. Techniques for economic valuation to deal with all forest goods and services need further development, such as choice modelling methods.

This analysis suggests that immediate effort should focus on removing those economic incentives which currently encourage forest loss and degradation. The development of markets for forest goods and services will be important, especially for carbon storage and sequestration and, on a more local scale, for tourism and sale of genetic material. Establishing clear, enforceable and transferable property rights for individuals or communities is likely to be an important precondition for sustainable long-term conservation and use. Mechanisms are also needed to ensure that the situation of those who receive the benefits from forest goods and services is altered in some way that will compensate those who bear the costs. Some encouraging examples are now being developed. However, the limitations of market mechanisms also need to be explored and recognized in relation to the needs of stakeholders, for example indigenous and local communities. Market mechanisms must complement other mechanisms, including legislation, regulation, certification, capacity-building, and addressing wider underlying causes (see the following sub-section).

IV. Causes of loss of forest biological diversity

Since there is a clear relationship between deforestation and loss of biological diversity, in order to identify and propose measures aiming to halt and reverse the loss of global forest biological diversity, the direct and underlying causes of forest decline should be addressed. Effective local action will specifically require a detailed understanding of the causes of loss of forest biological diversity

Due to the current national and global policy and economic frameworks and mechanisms, it is presently cheaper to log forests in an unsustainable way than to manage them sustainably. This factor has been identified in the present report as one of the primary causes for the high rate of deforestation and forest degradation, and therefore for the current loss of forest biological diversity.

Forest decline and/or loss of associated biological diversity result from many direct causes, some of which are natural but are aggravated by humans, such as climate change. The most important factors are human-induced causes, including conversion to agricultural land, dismantling of agro-forestry systems, overgrazing, unmitigated shifting cultivation, unsustainable forest management including poor logging practices, over-exploitation of timber, illegal logging, fuelwood and charcoal, over-exploitation of non timber forest resources - including bush meat and other living organisms - introduction of alien and/or invasive plant and animal species, infrastructure development (road building, hydro-electrical development, improperly planned recreational activities, urban sprawl), mining and oil exploitation, forest fires caused by humans, and pollution.

The underlying causes of forest decline are the forces that determine, through complex causation chains, the actions of the primary actors. They originate in some of the most basic social, economic, political, cultural and historical features of society. They can be local, national, regional or global, transmitting their effects through economic or political actions, such as trade or incentive measures. They are both numerous and interdependent and the approaches to deal with them are country-specific and will therefore vary among countries. In analysing the increasing literature available on the subject, in particular the recommendations and proposals for actions of the Intergovernmental Panel on Forests (IPF), the Intergovernmental Forum on Forests (IFF) and the work of the Centre for International Forestry Research (CIFOR), the following main underlying causes of forest decline were identified:

(a) *Broader macroeconomic, political and social causes,* such as population growth and density, globalization, poverty, unsustainable production and consumption patterns, ill-defined and implemented structural adjustment programmes, political unrest and wars;

(b) *Institutional and social weaknesses,* such as lack of good governance, lack of secure land tenure and uneven distribution of ownership, loss of cultural identity and spiritual value, lack of institutional, technical, and scientific capacity, lack of information, of scientific knowledge and the use of local knowledge, in particular lack of awareness of the value of forest biological diversity for provision of goods and services;

(c) *Market and economic policy failures,* such as under-valuation of forest biological diversity goods and services; perverse incentives; and subsidies;

(d) *Other policy failures,* such as ill-defined development programmes, ill-defined or unenforced regulatory mechanisms, lack of clear environmental policies and of environmental impact assessments.

V. Policy developments

Conservation and forest-related policies have, most often, failed to significantly reduce forest decline. This is primarily due to the inability to address the underlying causes of deforestation and forest degradation. In many countries, the weakness of conservation and sustainable management efforts is largely due to poor governance, lack of political will, lack of clear land tenure and land use rights, lack of adequate valuation of forest biodiversity, lack of appropriate local and global economic environments, insufficient implementation capacity, lack of financial or human resources and of environmentally sound technologies.

Some positive elements are also evident, mostly in the area of forest policies and forest management practices. These have partly resulted from a series of international (IPF, IFF, United Nations Forum on Forests (UNFF)) and regional forest processes and initiatives around the world, addressing the development of sustainable forest management. National forest programmes are increasingly being developed as a means to address forest sector issues in a holistic manner, taking into account other relevant sectors which impact on forests. The importance of national forest programmes has been confirmed and stressed by IPF, IFF and UNFF. Due to increasing public awareness of biological diversity issues and of the goods and services forest ecosystems produce, there is increasing support for sustainable forest management among consumers, politicians and industry. A substantial section of the timber trade seems to be prepared to take environmental issues seriously and to make real efforts to change its practices. Although such positive trends do not yet seem to have substantially influenced the loss of forest biological diversity, it is possible to note some of them that may contribute in some way to the maintenance of forest biological diversity:

(a)Development of national forest programmes

(b) Increased number and area of protected forest areas

(c)Development of improved ecological forest management and forestry practices, including landscape ecological planning procedures, identification and preservation of key biotopes and other key elements in the forest landscape, “reduced impact” logging, and “close to nature” forestry

(d)Mechanisms to illustrate sustainable forest practices such as demonstration forests (e.g. International Model Forests Network initiatives)

(e)Many initiatives on criteria and indicators for sustainable forestry

(f)Independent certification of sustainable forest management and related labeling of forest products originating from well-managed forests.

There is also an increase in the willingness to accept issues related to the rights, needs and participatory possibilities of indigenous people and local communities in the context of forest conservation and management. This positive development includes donor institutions’ interest in collaborating directly with indigenous and local communities, policy revision by many relevant actors, and increased acceptance of traditional knowledge and collaborative management in forest conservation and management. However, these changes have happened largely at the international level and often have not been sufficiently incorporated into national policies.

Difficulties in addressing relevant socio-economic issues in the context of forest biological diversity are also related to poor knowledge. Present knowledge concerning the use and valuation of non-timber forest products and services, the cultural and spiritual values of forests, or the development of the rights, needs and participatory possibilities by indigenous people is sparse or receives inadequate attention.

VI. Conclusions

The Expert Group has drawn a number of key conclusions from the review of information summarized in sections A to E above:

(a)Forest issues relate to a range of political, economic, social, cultural, environmental and scientific aspects, which must be dealt with in a coordinated, cross-sectoral and holistic way;

(b)Assessing the current global status of forest biological diversity in quantitative as well as qualitative terms is problematic, because of the difficulties quantifying biological diversity. An immediate need exists to categorize and substantially improve the understanding of biological diversity, with a view to measuring trends, particularly on regional scales;

(c)The rate of deforestation has been at a high level for many centuries and continues to be at a dramatically high level, with most current deforestation occurring in tropical forests;

(d) Large-scale degradation of forest quality occurs in all regions and forest types due to human activities, and this is exacerbated by improved access to intact forests;

(e) The number of extinct and endangered forest species, already at historically high levels, can be expected to rise due to an existing "extinction debt" and the continued habitat loss, fragmentation, invasive species and over-exploitation. The evidence also clearly shows that an "extinction debt" exists, i.e. many extinctions will happen in the future as a result of the deforestation and degradation which have already occurred;

(f) Plantations have a role to play in conserving and enhancing forest biological diversity, but cannot compensate for deforestation of primary forest and consequent loss of particularly rich biological diversity;

(g) There is a clear need to better monitor and report changes in the quality and quantity of the world's forests, from the national to the global level;

(h) Data on rare and threatened forest species alone are insufficient to provide a reliable picture of broader trends in biological diversity. Surrogates for biological diversity, such as umbrella species, indicator species, key habitats and structural indicators, may help to assess and predict the effectiveness of conservation and forest management programmes and should be included in sustainable forest management criteria and indicator lists;

(i) There is generally less knowledge with respect to forest biological diversity in tropical forests compared to the other two biomes;

(j) Adequate attention needs to be given to the principles, methods and ways and means for the potential use of traditional knowledge of indigenous peoples and local communities as a valuable tool for forest biodiversity management;

(k) Protected forest areas have increased in recent years in both number and area. However, globally, forest types are neither well protected nor well represented in protected areas. The pattern of protected forest areas remains uneven, especially in terms of distribution and the representativeness of many forest types. The effectiveness of the protection provided in protected areas remains a major problem;

(l) It should be recognized that the conservation of forest biological diversity should be an overall objective of sustainable management of all types of forests by all countries, and not be limited to protected forest areas;

(m) The relationship between biological diversity and ecosystem goods and services is direct, but the exact linkages remain unclear and require research. Critical levels of biological diversity loss and/or change, as well as the human impacts that cause them and which affect forest ecosystem functioning and forest goods and services, are still largely unknown;

(n) The implementation of the ecosystem approach should be the overarching framework for sustainably managing forests. In particular, the ecosystem approach requires adaptive management to deal with the complex and dynamic nature of forest ecosystems and the absence of complete knowledge or understanding of their functioning. As a consequence, forest ecosystem managers should consider the effects – actual or potential – of their activities on forest ecosystems and take into account that forest ecosystems should be managed within the limits of their functioning. In this respect, conservation of forest structure and functioning should be a priority target;

(o) To help to implement the ecosystem approach, research must address an understanding of the effects of forest management on biological diversity on all scales, from genes to landscapes, to provide a basic understanding of the role of biological diversity in forest functions and processes. Monitoring of forest biological diversity and of the changes caused by forest management is important in order to assess the effectiveness of management strategies and the cumulative change of forest use;

(p) Restoration of forest biological diversity in degraded forests and deforested lands is an issue of growing importance in both the developed countries and the developing world. There is a need to focus more on the potential for synergy from combining different forest categories, including primary and secondary natural forests, agro-forests and new forest plantations, to achieve a specified range of forest biological diversity and related goods and services. The means to restore forest biological diversity in different situations are poorly understood and research should be increased in this area;

(q) Current economic incentives often encourage forest loss and degradation and are therefore disincentives to sustainable forestry;

(r) Sustainable forest management is generally less profitable in monetary terms than ecologically non-sustainable forest practices. Local and indigenous communities and, ultimately, nations are likely to be the major losers from the conversion of forested land to other uses and non-sustainable forest practices;

(s) There is a need for more effective participation of the inhabitants of forests, indigenous peoples and local communities in all processes related to forest use and management. A stakeholder analysis at all levels, from local to global, would be a valuable basis for discussions and decisions on forest biological diversity use and its management;

(t) Effective action for forest biological diversity needs to address both the direct and the underlying causes of loss, and this requires a more detailed understanding of these causes at both the international and national levels, since each country has different circumstances and will need a specific approach. Many of the issues can only be addressed globally or regionally;

(u) The underlying causes of loss of forest biological diversity are very fundamental and complex and they derive from broader macro-economic, political and social causes, such as poverty, rapid population growth, globalization of trade, unsustainable production and consumption patterns, political unrest, lack of good governance, land rights disputes and lack of institutional technical and scientific capacity. Loss of forest biological diversity cannot be stopped and reversed without addressing these and other fundamental problems; as well as improving our knowledge of biological diversity and developing more sustainable forms of forest management;

(v) Many of the threats to forest biological diversity emanate from non-forest sectors, such as agriculture, land use, industry, energy and others. The development of cross-sectoral linkages, e.g. through a consistent development of national biodiversity strategies and national forest programmes, possibly within the framework of national sustainable development strategies, is therefore very important;

(w) Present knowledge concerning the use and valuation of non-timber forest products, the cultural and spiritual values of forests, or concerning the developments of rights and participatory possibilities by indigenous people is sparse and needs more adequate attention.

There are some positive trends and developments upon which to build, mainly in the area of improving forest policies and sustainable forest management practices, which include biological diversity provisions. Public and consumer awareness is leading to the development of a more serious interest in biological diversity and environmental issues by other stakeholders, including politicians and the private sector. The promotion of certification of forest products, when done properly, can also be an encouraging development to provide positive incentives for sustainable forest management.

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I. STATUS OF FOREST BIOLOGICAL DIVERSITY

A. DISTRIBUTION OF THE WORLD'S FORESTS

The area[1] of the world's forests (see Annex I for use of terms), including natural forests and forest plantations, was estimated to be 3869 million ha in 2000, equivalent to almost 30% of the ice-free land area of the earth (FAO, 2001a). Loss of natural forests is estimated to be 16.1 million ha annually, with 15.2 million ha of this occurring in the tropics (FAO, 2001a). This loss is not compensated for by regrowth of natural forests or plantations, leading to a net annual loss of 9.4 million ha of forests. The three major forest biomes are boreal, temperate and tropical. In terms of area, the forests are almost equally divided between tropical/sub-tropical forests and temperate/boreal forests (See Map 1). The original and current forest cover of the Earth is given in the map 2.

The majority of the forested area consists of natural forests (95%), with commercial plantations comprising 3% and other forest plantations making up the remaining 2% (Carle *et al*., 2001; FAO, 2001a). Under the FAO (FAO, 1998) definition, natural forests include all forests "composed of indigenous trees, not planted by man or in other words, forests excluding plantations", while plantations include "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". Detailed information on the world's forests, at regional and national levels, is given in FAO (1997, 1999, 2001a, 2001b). A little over half (55%) of the world's forests are located in developing countries. Two-thirds are found in only ten developing countries. Brazil has 544 million ha, Indonesia 105 million ha, Democratic Republic of Congo 135 million ha, Peru 65 million ha, India 64 million ha, Mexico 55 million ha, Bolivia 53 million ha, Colombia 50 million ha, Venezuela 50 million ha and Sudan 42 million ha. More than three quarters of the temperate and boreal forests are situated in just four countries: Russian Federation 851 million ha, Canada 245 million ha, USA 226 million ha and China 163 million ha. Global forest cover in 2000 for different regions is given in Table 1 (from FAO, 2001b).

Table 1. Regional forest cover (FAO, 2001b)

Region	Land area	Total forest 2000	
	Million ha	Million ha	%
Africa	2 964	650	17
Asia	3 085	548	14
Oceania	849	198	5
Europe and Russ. Fed.	2 260	1 039	27
North & Central America	2 137	549	14
South America	1 752	886	23
World Total	**13 048**	**3 869**	**100**

Table 2. Total forest area by ecological zone and distribution between regions, according to FRA 2000 global ecological zoning and global forest cover map.

Distribution of percentages do not exactly tally with other area statistics due to systematic distortions in the remote sensing classification of forests in the global map.

[1] Although there are caveats to the forest data provided by the FAO in the Forest Resource Assessment 2000 based on definitions, sampling techniques and unclear linkages to forest biological diversity (Stokstad, 2001; Matthews, 2001), these data provide a necessary background for understanding global forest change and decline.

Ecological Zone	Total forest	Africa	Asia	Oceania	Europe	North & Central America	South America
	million ha	%	%	%	%	%	%
Tropical rainforest	1090	24	17			1	58
Tropical moist deciduous	410	40	14	6		9	31
Tropical dry	180	39	23			6	33
Tropical mountain	150	11	29			30	30
Subtropical humid forest	170		52	8		34	6
Subtropical dry forest	30	16	11	22	30	6	14
Subtropical mountain	130	1	47		13	38	1
Temperate oceanic forest	30			33	33	9	25
Temperate continental forest	270		13		40	46	
Temperate mountains	130		26	5	40	29	
Boreal coniferous forest	730		2		74	24	
Boreal tundra woodland	130				19	81	
Boreal mountain	410		1		63	36	
TOTAL	3869	17	14	5	27	14	23

A good overall picture on World's forests can be obtained on basis of three maps representing various aspects of the global forest cover. Map. 1. *Forests 2000 by major ecological domains* is published on the web site for the FAO Forest Resources Assessment 2000 (http://www.fao.org/forestry/FO/fra/index.jsp). The map presents the world's forests according to the main biomes, and classified as a) closed forests, b) open and fragmented forests, and c) other wooded land. Map. 2. *Global distribution on current Forests* by the UNEP-WCMC (http://www.unep-wcmc.org/forest/global_map.htm) gives the global distribution on 22 main classes of natural forests and 5 classes of plantations. Map. 3. *Original and current Forest Cover* by the UNEP-WCMC (http://www.unep-wcmc.org/forest/original.htm) shows the estimated situation between the end of the last Ice Age and the expansion of human activities, about 8,000 years ago. This map reveals large losses in the global forest cover since this historic baseline, as well as great regional differences in the deforestation.

Protected forested areas

At the global level about 30,350 protected areas have been established, covering 8.8% of land area (IUCN, 1998). Green and Paine (1997) have endeavoured to estimate the extent to which the major biomes, including various categories of forest[2], are represented overall in the global protected areas network[3] (see Table 3). In this analysis, tropical forest types are better represented in protected areas than temperate forest types, mainly due to more extensive deforestation over a longer period in temperate regions of Eurasia. The overall figures for tropical forests appear satisfactory, approximating the 10% protection target established at the IV Worlds Parks Congress (IUCN, 1993), but in reality overestimate the extent to which forest ecosystems are being properly conserved in protected areas. A recent survey of 10 developing countries with major forest resources found that only 1% of forest protected areas are secure in the long-term, with 60% currently secure but with threats likely in the near future and more than 20% suffering from degradation, (Dudley and Stolton, 1999). An high proportion (59%) of protected areas are less than 1000 ha in size, and their long-term viability for conservation of forest biodiversity is open to question unless they are located near and linked with other forested protected areas and are

[2] Assessing forest ecosystem diversity requires meaningful systems of classification of forests. Useful systems have usually been developed at national and regional levels, but are increasingly difficult to generate as the geographical scale increases, and are problematic at the global scale. FAO's Forest Resource Assessment 2000 (FRA 2000) is addressing this issue through a combination of remote sensing data and eco-regional classifications.

[3] This particular analysis collectively under-represents the protection of biomes by about 30% because information was not available on the particular biome represented in many protected areas.

subjected to comprehensive protection and management regimes. Furthermore, it is readily apparent that data on the extent of conservation of different, very broadly classified, forest types or biomes provide only limited insight into how well forest biological diversity (FBD) is being conserved. Only when vegetation communities (forest associations or types) have been thoroughly surveyed and mapped at a sufficiently detailed resolution, e.g. a minimum mapping unit of less than 100 ha (Jennings, 1994) and at a minimum of 1:50,000 mapping scale, will it be possible to ascertain the extent to which different forest types are represented within the protected areas network, and to ascertain the exact contribution of the protected areas system to the overall conservation of FBD.

Table 3. Protected forested area as a percent of the total forest area for various global regions (source: WCMC, 2000[4];).

Region	Forest area (10^3 km^2)	Protected forest area (10^3 km^2)	% Forest protected
Africa	5,683.1	496.9	8.7
Australasia	1,493.2	125.8	8.4
Caribbean	53.8	7.9	14.7
Central America	902.0	88.1	9.8
Continental SE Asia	1,707.7	192.5	11.3
Europe	1,815.4	144.8	8.0
Far East	1,456.0	77.4	5.3
Insular SE Asia	1,468.4	247.5	16.9
Middle East	1,676.7	6.4	3.8
North America	8,454.0	700.0	8.3
Russian Federation	8,257.1	150.6[5]	1.8
South America	8,429.5	874.9	10.4
Total	39,887.9	3,112.8	7.8

4 http://www.wcmc.org.uk/forest/data/cdrom2/gtabs.htm

5 According to Filipchuk 2001 amount of protected nature area is about 5% of the Russian forest area. The State Natural Reserves (zapovedniks) totaled 33.5 million hectares at the beginning 2001.

B. STATUS OF BIODIVERSITY IN FOREST BIOMES

The intent of the following sections is not to be exhaustive in the review of knowledge about biological diversity, particularly within biomes. For substantial detail, the “Global Biodiversity Assessment” published by UNEP (1995) is recommended. The intent here is to focus on key issues that lead to a better understanding of the status of forest biological diversity, as well as to identify gaps in knowledge. This approach leads toward recommendations of options to improve the conservation and the sustainable management of forest biological diversity.

Forest biological diversity can be quantified at several scales, these include: assessing the genetic diversity within species, counting the number of species per unit area (local, regional, national, continental, global), determining numbers and arrangement of forest types and their age, classifying types of forest ecosystems, determining communities of species associated with forest ecosystems and describing landscape structure.

i. Boreal forests

Distribution and structure

Boreal forests, including tundra woodlands, extend over about 1270 million hectares, or about one third of the world’s forest cover. The boreal forest is the second largest terrestrial biome after tropical forests. This northern circumpolar biome is strongly characterised by coniferous ecosystems with low tree species richness, extensive and fairly uniform stands and relatively short-lived species (<200 years), which are under fire**,** wind and insect disturbance regimes. Extreme oceanic types with broad-leaved deciduous trees are found in northwestern Europe, where the tree limit is formed by *Betula pubescens* subsp. *czerepanovii*. Similar ecological conditions prevail in northern Asia, Alaska, and north Canada, with stunted *Picea*, *Larix, Pinus pumila* and *Betula nana* at the treeline.

Boreal landscapes are composed of a complex of plant communities that, aside from vast tracts of forest stands, include various wooded and open mires or bogs, numerous water bodies of varying size, rivers, rock outcroppings and natural grasslands and fens (Walter, 1979; Barbour and Christensen, 1993). Local distribution of boreal communities is heavily influenced by topography, hydrology and soils. Spodosoils or podsols dominate upland sites. Glacial features such as eskers, moraines, kettle holes and outwash plains create considerable variation in the landscapes. Paludification can be very extensive, especially in northwestern Europe and Russia, western Siberia, and the Hudson Bay lowlands and central Keewatin, District of Canada. The extensive continental larch forests of eastern Siberia, on permafrost areas with thermokarst formations, show a great diversity of habitats. Most of the conifer species are adapted to regenerate following fire either as a result of serotinous cones, or as a result of their shade-tolerant development beneath a mixed or deciduous canopy. Some species cannot regenerate without this canopy, as they need it for protection from frost and excessive sunlight. The boreal biome is characterized by an extensive dormant period, often of extreme cold, which reduces its annual productivity. Net annual primary production on upland sites varies from 400 g/m^2 on poor sites to 1300 g/m^2 on mesic fertile sites (Barbour and Christensen, 1993).

In North America, regional classifications of the boreal forest ecosystems (e.g., Sims *et al.*, 1989) are available and collectively define over 500 forest ecosystems, mostly based on understory species. Many boreal plant and animal species have particular habitat requirements. The fire regime and gap-phase dynamics of these forests provide a set of hydrological, edaphic and microclimatic conditions that create a wide array of habitats. The boreal forests also have structural components that are important to the fauna within them These include uneven canopy structure, size and age variation of the trees, especially the occurrence of tall, old coniferous and deciduous trees, stumps of trees and standing and fallen decaying wood of various sizes.

The southernmost cool temperate, oceanic forests found along the western coast of South America and New Zealand are considered to be a southern hemisphere equivalent of boreal forests. These forests, rich in epiphytic cryptogams, are dominated by both evergreen and deciduous broad-leaved *Nothofagus* species, while mires or bogs are also characteristic (Walter 1979).

In the boreal forest regions, an average daily temperature of more than 10°C occurs on less than 120 days and the cold season lasts longer than six months. The northern transition from boreal forest to tundra (barren lands with frozen subsoil) is related to a short growing season (fewer than 30 days with a daily temperature above 10°C) and permafrost conditions unfavourable to tree growth (Walter, 1979). The ecozone between boreal forest and tundra, called forested tundra, is characterised by stunted trees and shrubs and may extend hundreds of kilometres on flat terrain. At its southward extremes, the boreal biome reaches areas where the climate is sufficiently favourable to support deciduous broad-leaved species more typical of temperate forests. The zone between the coniferous boreal forests and the temperate deciduous broad-leaved forests often consists of mixed forests, recognised as a hemiboreal or boreonemoral or transitional zone. The climate in the vast *taiga* zone varies widely from cold, oceanic with a relatively small temperature amplitude, to cold continental in which, in extreme cases, an annual temperature span of 90°C may occur. Temperatures also change from north to south, and several subzones can be distinguished: northern, middle, southern and extreme continental.

Species diversity

The Wisconsin glacial events, 10-14,000 years ago, forced plant and animal life further south, followed by northward migration in recurrent cycles. The boreal forest biome is distributed across areas formerly covered by continental glaciers and, consequently, the land has supported forest cover for only 3,000 to 7,000 years (Ritchie 1987). The number of tree species that characterise these forests is therefore low, especially in the Euro-Siberian area, where major watercourses and mountain ranges run at right angles to the direction in which the species migrated northwards. As a result of the post-glacial history of the biota, many boreal and subarctic tundra species have wide distributions. There are relatively few endemics at the species level, and most of these occur in the extreme eastern and western parts of the continents, close to ancient refuges. Due to wide distributions and varying environmental conditions, evolution at the level of ecotypes and subspecies is common and some genera, such as *Salix*, *Carex* and *Betula,* show wide-scale hybridisation (Jonsell, 2000).

Boreal forest stands normally contain no more than a few species, primarily of the genera *Picea, Pinus, Abies, Larix, Thuja, Betula, Prunus, Alnus* and *Populus,* and they often form monocultures, particularly in the case of *Picea*, *Pinus* and *Larix*. These genera are panboreal and members of the four deciduous genera (*Betula, Prunus, Alnus* and *Populus*) grow more rapidly than the conifers and tend to occupy sites immediately following stand disturbance. Tree species richness in North American forests is greater than in the Euro-Siberian region. In North America, four of the six principle boreal forest species extend across the continent, though no single tree species is panboreal. *Picea mariana* grows on poor soils and forms the northern treeline continent-wide. Where fire is uncommon, *Abies* spp. often predominate in the eastern and continental North American boreal zone. In Eurasia, this genus is ecologically largely replaced by two species of *Larix*. Larch forests, mostly consisting of *Larix gmelinii*, cover 2.5 million km^2 in continental Siberia where much of the terrain has deep permafrost. *Larix sibirica* often forms monotypic stands following disturbance by fire (Schulze *et al.*, 1996), while in North America, *Larix laricina* is rarely a dominant species, it is found mainly on cold, wet and poorly drained sites such as in sphagnum bogs and muskeg. In Europe, only *Picea abies* and *Pinus sylvestris* are true dominants of the boreal zone, and are often mixed in successional phases with broad-leaved deciduous tree species such as *Betula pendula, B. pubecens, Populus tremula* and *Alnus glutinosa* and *A. incana*. In more eastern European regions, *Picea abies* is replaced by the closely related *Picea obovata*, with *Abies sibirica, Larix sibirica* and *Pinus cembra* subsp. *sibirica.* There is a broad belt of hybrids, *Picea abies* x *P. obovata*, between their natural regions. In Eurasia, the proportion of *Picea* gradually decreases eastward while that of *Larix* increases correspondingly. In northern Japan, the number of coniferous species increases again.

Conifers comprise the bulk of the biomass in these boreal ecosystems, although most forests also include a variety of deciduous tree and shrub species, dwarf-shrubs (notably members of the Ericaceae), grasses, sedges and herbs. In general, species diversity in taiga communities increases with the length of the growing season, increasing soil fertility and favourable drainage. A comparatively moderate richness of bryophytes, lichens and fungi occur in many boreal forest types, they are especially common in older forests with their greater volume of decaying wood.

Plantations in the boreal biome

Plantations are often used as a silvicultural technique in boreal forests, replacing a harvested stand with species from the region, although the species mix may not be exactly the same as was previously on the site. In particular, conversion from mixed deciduous-coniferous forests to conifer plantations is common. About 25-60% of boreal stands that are logged are subsequently replanted or seeded, depending on the country. Among the boreal forest countries, Sweden and Finland have the most intensively managed plantations, The major distinction between plantations in boreal forests and those of the temperate or tropical biomes is that, in the boreal biome, there is minimal planting of exotic species and the plantation forests maintain much of the biodiversity that originally occurred at the site. Perhaps the major impact of plantation forestry in the boreal biome results from harvesting the stands before they develop old forest characteristics such as cavity trees and fallen woody debris. These planted forests are not included in the FAO statistics on plantation forests (FAO, 2001a).

ii. Temperate forests

Distribution and structure

The temperate forest biome, located in the mid-latitudes, occupies a climatic zone with pronounced variations in seasonal temperatures, characterised by distinct winter and summer seasons, with a daily mean temperature over $10^{o}C$ for more than 120 days (Walter, 1979)**.** This biome occurs primarily in the northern hemisphere, while in the southern hemisphere, it is limited to the southern part of the Andes in Chile and in portions of New Zealand, South Africa and southeast Australia. Temperate forests are dominated by deciduous tree species and, to a lesser extent, evergreen broad-leaf and needle-leaf species (Melillo *et al.*, 1993). More than 50% of the original temperate forest cover has been converted to agriculture (Matthews, 1983). Unfortunately, most forest statistics do not distinguish between natural forest, secondary forest and plantations. Occurrence of temperate forests is highly concentrated in the Russian Federation, Canada and the United States, which together have over 70% of the total, with the Russian Federation alone holding over 41% of the world's temperate forests. However, from an ecological perspective, some of the smaller temperate forest areas are critical sources of biological diversity, including, for example, those in parts of Europe, Australia, South America, the remnant temperate forests of South Africa and geographically isolated and highly endemic natural forests of New Zealand.

In Europe, temperate forests extend over some 160 million ha, which represents slightly less than half of the original forest cover. In western Europe (Matthews, 1983), it is estimated that the extent of remaining old growth and semi-natural forest is only 0.8% of the original forest cover (Ibero, 1994). Eastern Europe has more old growth forest than in the west (Ryzkowski *et al.*, 1999). In the United States, less than 2% of the original temperate forests remain, although proportions vary regionally. For example, the states of Washington and Oregon have 13% old growth temperate forests remaining. In British Columbia, Canada, almost 40% of the original natural forests remain, although some of these are subject to intensive forest management (Canadian Council of Forest Ministers, 2000). New Zealand retains less than 24% of its of its native forests (Clout and Gaze, 1984) and in Australia, the amount of the original temperate forest varies from 5-20%. In some temperate areas of developing countries, there is a net loss of forest cover; Chile, for example, loses about 20,000 ha/year [FAO, 2001a/2001b].

The annual productivity of natural northern temperate forests is about 900 to 1000 g/m^2 and up to 1000 to-1400 g/m^2 in old southern temperate forests of North America (Lieth and Whitaker, 1975). However, there is obviously a large variation associated with these figures depending on site, elevation, type and age of forest.

Mediterranean forests constitute a distinct sub-zone of the temperate biome and occur between 30 and 40 degrees latitude on the west and south-west coasts of the continents. Their climate is characterized by hot, dry summer and mild, moist winters. The Mediterranean sub-zone in the Americas occupies coastal California in the United States and the coastal region of Chile. In Africa, these forests extend around the Cape of Good Hope; they also occur in the southern part of Australia. However, the largest Mediterranean sub-zone is located around the Mediterranean Sea and includes the southern part of Europe, the south-west part of Asia and the north coast of Africa. In Europe, the Mediterranean sub-zone has been the cradle of several civilisations, one replacing another over centuries, and this has resulted in a long history of extensive environmental change as a result of economic, cultural and social activities. The area surrounding the Mediterranean Sea was originally covered with forests of *Cedrus libani*, *Quercus ilex, Quercus cerris, Arbutus unedo, Pinus halepensis* and, *Pinus nigra*, but the Mediterranean hillsides were transformed hundreds of years ago into terraces of fruit orchards, gardens, olive tree and fig tree plantations, as well as human settlements. Areas that have escaped cultivation are covered with shrubs and bushes, resulting in *Maccia (maquis),* a woody secondary vegetation cover (Ovington, 1983).

Species diversity[6]

More than 1200 tree species are represented in the temperate biome (Ovington, 1983; Schulze *et al.*, 1996). Globally, temperate deciduous forests maintain a large variation in species richness, resulting largely from climate and differences in geological history. During the Tertiary period (3 million+ years ago), the three deciduous forest regions of the northern hemisphere are thought to have had a fairly uniform tree flora. Europe and North America were still closely related floristically and there were also many common species in Europe and Asia (Walter and Straka, 1970). However, during the Pleistocene glaciation, the east to west orientation of mountain systems, such as the Alps, the Caucasus and the Himalayas, apparently formed a barrier, resulting in the Euro-Siberian flora being reduced as many species could not survive the cold in various small refugia. However, in North America, the mountain chains are oriented north to south, enabling easy migration, so most species survived the glacial periods in southern locations (Ritchie, 1987). The highest temperate species post-glacial survival, and hence current diversity, is in Asia (Ohsawa, 1995), with four times the number of tree species there than in North America (Huntley, 1993).

East Asia's forests are very rich in woody plant species, with almost 900 trees and shrubs. That is almost six times greater than in North America, where the second most diverse temperate forests occur. The temperate forests of Europe are more impoverished, with just 106 tree species and significantly fewer families and genera than in North America. The southern hemisphere generally has even fewer species than Europe (except for Australia with its high diversity of *Eucalyptus* and *Acacia* species), but there is a high endemism with most species belonging to different families from those found in the northern hemisphere, suggesting major differences in evolutionary history. Transition zones between tropical and temperate forest biomes, are comparatively species rich. These occur, for example in Japan and the southern United States where temperate lowland forests merge with subtropical evergreen broad-leaf forests. In southern Canada, the maximum tree species richness in temperate forests is approximately 60 species, but by mid-latitudes in the eastern United States, the same biome contains over 100 tree species, illustrating the general latitudinal relationship of species diversity, i.e. diversity increasing towards the equator (Stevens, 1989).

Temperate forests tend to support their largest variety of species on nutrient-rich soils, and species richness also seems to be greater on alkaline and neutral soils than on acid soils (SCOPE, 1996). Local

[6] Much of the discussion of temperate forests is taken from Ovington (1983) and Schulze *et al*. (1996) and these serve as references for all statements.

species richness in many of these forests is highly variable, ranging from monocultures to multi-species forests. In many areas of the temperate biome, large stands of deciduous forests may be composed of a single tree species. For instance, *Fagus sylvatica* dominates deciduous forests in Europe, *F. orientalis* forms nearly pure stands in the montane region of the Caucasus, and *F. crenata* is predominant in pure stands in the wetter regions of Japan. In Europe, on calcareous soils with high watertables, *Quercus* and *Carpinus* become dominant rather than *Fagus*. In North America, *Fagus* rarely dominates forests, but pure stands of *Betula* and *Populus* are common, as is the case in Siberia and northern Japan. *Nothofagus* occurs in monocultures in New Zealand and South America. *Quercus* and *Pinus* are global species found in most northern hemisphere temperate forests. In Australia, forests are dominated by the extremely diverse genus *Eucalyptus* with more than 70 species in 16 forest types (Ovington and Pryor, 1983) whereas *Quercus* is absent. Although alpha-diversity (patch-scale or within-site diversity) may be low, beta-diversity (regional or among-site diversity) in the temperate biome forests can be quite high.

In North America, an important temperate coniferous forest belt occurs along most of the west coast from Alaska southwards to northern California. The forests lie on the windward side of the coastal mountain chain, which runs the length of the continent. These forests, collectively referred to as temperate rainforests, exhibit a high level of biological diversity with a large number of endemic plants and animals (Ruggiero *et al.*, 1991). They are characterized by several long-lived tree species (>1000 years) and contain the tallest trees in the world (to 95 m), including: *Sequoia sempervirens*, *Sequoia gigantea*, *Pseudotsuga menziesii*, *Picea sitchensis*, *Tsuga heterophylla*, *Thuja plicata* and *Chamaecyparis nootkatensis* (Maser, 1990). These trees are particularly valuable to the forest industry. Trees of this size and ecosystems as complex as these occur nowhere else in North America (Maser, 1990). The management of the temperate rainforests forests has generated more controversy than that of any of the other North American forest types because of their species diversity, complex functioning and the particularly majestic characteristics of the old-growth trees, which can exist for many centuries in a gap-phase dynamic condition.

As with boreal forests, the fauna of temperate forests, especially the birds and mammals, can have a wide distribution and even extend to other biomes. For example, Neotropical migrant birds of North America, numbering about 250 species, make the annual trip from the tropics to the temperate regions, and changes in the extent and condition of either forest biome can affect the populations of these birds in both continents. Survival of these birds is important because smaller numbers may allow defoliating insects to reach epidemic proportions more frequently and this further endangers the survival of some species (UNEP, 1995). Not all temperate forests host fauna with such a wide distribution. In the forests of southern South America, South-East Asia, Australia and New Zealand, there are many endemic species of mammals and birds that are highly localized (WCMC, 2000).

More animal species have become extinct in the past 1000 years, or have had their range and population substantially reduced, in the temperate forest biome than in the other biomes (Hilton-Taylor, 2000). Falling particularly into this category are the large ungulates including extinct aurochs *(Bos taurus*) and tarpan (*Equus gmelini silvaticus),* endangered bison (*Bison bonasus)* and declining fallow-deer *(Cervus dama*) and moufflon *(Ovis musimon*) in eastern Europe. The general reduction of forest cover, combined with hunting and/or trapping, has caused the reductions of many large carnivores such as the brown bear (*Ursus arctos*), lynx (*Felis* spp.), cougar (*Puma* spp.), glutton or wolverine (*Gulo* gulo) and wolf (*Canis* spp.) (Hilton-Taylor, 2000; Pimm *et al.*, 1995). Within the past 200 years in North America, the passenger pigeon (*Ectopistes migratorius*), Carolina parakeet (*Cornuropsis carolinensis*), ivory-billed woodpecker (*Campephilus principalis*), Bachman's warbler (*Vermivora bachmanii*) and the eastern cougar (*Puma concolor*) have become extinct (Pimm *et al.,* 1995).

Plantations in the temperate biome

Plantations make up 5% of the world's forests (FAO, 2001b), though only 3% when considering commercial plantations. According to the FAO Forest Resource Assessment (FRA, 2000), forest plantations cover 187 million ha, of which those in Asia account for 62%. The largest plantation resources are found in China (24%) and India (17%). The area of forest plantation has increased significantly compared to the 1995 estimate of 124 million ha. The new annual planting rate is 4.5 million ha globally, with Asia and South America accounting for 89% of that. Broad leaf species account for 40% of global forest plantation resources, conifers 31% and not specified 29%. Globally, 48% of the forest plantation estate is for industrial end-use, 26% for non-industrial end-use (fuelwood, soil and water conservation, etc) and 26% is for unspecified use. Forest plantation ownership is 27% public, 24% private, 20% other and 29% not specified (Carle et al., 2001).

Plantations are often locally and regionally important in influencing floral and faunal richness (Peterken *et al.*, 1992). The recent dramatic increase in the area occupied by forest plantations will have an increasingly important long-term effect on biodiversity. In many areas of the temperate biome, human activities have considerably influenced forest stands and landscape structure. This is especially true for countries of Europe and Asia, with their long history of habitation, where human use of forests has reduced diversity and many stands may now have fewer than 10 species of trees and shrubs. Conversion of broad-leaf forests into coniferous plantations began in Europe more 120 years ago and is still common today. For example, the forest cover of Germany has changed from what was formerly more than 90% broad-leaf forest (mostly *Fagus sylvatica*), into about 80% coniferous plantations (largely *Picea abies*). In a similar manner, much mixed deciduous forest in Japan has been converted into *Cryptomeria japonica* plantations in the south of the country and into *Abies* and *Picea* plantations in the north. Natural *Nothofagus* forests in Chile have been replaced by plantations of *Pinus radiat*, and the practice continues today (Gajardo, 1994). [Possible addition: In New Zealand planted forests of *Pinus radiata* began to be established during the first decades of the 20th Century. Production from these planted forests has largely replaced that from the podocarp, hardwood and beech (*Nothofagus)* natural forests enabling conservation of these natural forests] In North America, mixed forests were often converted into *Pinus taeda* plantations in the south, and *P. banksiana* and/or *P. resinosa* in southern Canada and the northeastern USA. In Europe, large areas of the temperate native forests have been replaced by plantations of introduced tree species, such as *Pseudotsuga menziesii*, *Picea sitchensis* and *Pinus contorta*. Many new plantations in European countries are established by planting species from non-local provenances to enhance wood productivity (Ryzkowski *et al.*, 1999).

The potential for forest plantations to partially meet demand from natural forests for wood and fibre for industrial uses is increasing. Although accounting for only 5% of global forest cover, forest plantations were estimated in the year 2000 to supply about 35% of global roundwood, with an anticipated increase to 44% by 2020. In some countries, forest plantation production already provides the majority of industrial wood supply.

iii. Tropical forests

Distribution and structure

In the tropical forest biome, three major regions are recognized: American, African and Indo-Malaysian-Australian (Whitmore, 1984). Tropical forests may be broadly classified as moist or dry, and further broken down into rain forest (some 66% of the total tropical moist forests), cloud forest, evergreen seasonal forest, dry evergreen seasonal forest, semi-evergreen tropical forest, moist deciduous forest (monsoon forest), dry deciduous forest, and mangrove. Rain forests occur in Central and South America, Africa, the Indo-Malayan region and in Queensland, Australia. Where several dry months (60 mm rainfall or less) occur regularly in the tropics, monsoon or seasonal forests are found. Both rain and monsoon/seasonal forests (closed forests) have together been termed ‘tropical moist forest’ (Sommer, 1976). Cloud forests situated at middle to high altitudes derive a significant part of their water supply from cloud and fog and support a rich abundance of vascular and nonvascular epiphytes. The evergreen seasonal forests are found in regions where every month is wet (100 mm rainfall or more) and in areas with only short dry periods (Whitmore, 1990). Dry tropical rain forests were originally described by Schimper (1903) as ‘evergreen, hygrophilous in character, at least thirty metres high, rich in thickstemmed lianes, and in woody as well as herbaceous epiphytes.' Mangroves are the characteristic littoral formations of tropical and subtropical sheltered coastlines, they have been variously described as 'coastal woodland', 'tidal forest' and 'mangrove forest'

Basing his work on previous classifications, Whitmore (1990) has, for convenience, grouped the formations within the tropical rain forest according to the main physical characteristics of their habitats, noting that the naming of vegetation types is always problematic. In this arbitrary arrangement, the first division is between climates with a dry season and those that are perhumid (for moist forest), the second division (for the rain forest) is a crude measure of soil water availability and distinguishes swamp from drier land forests. The third division is based on soils and, within dryland forests, distinguishes those on parent materials with atypical properties – peat, quartz sand, limestone, and ultrabasic rocks – from the widespread 'zonal' soils, mainly ultisols and oxisols. Finally there is a division of the forests on zonal soils by altitude. In the Indo-Malaysian region the tropical rain forest lies as a belt of evergreen vegetation extending through the Malay Archipelago from Sumatra in the west to New Guinea in the east (Whitmore, 1984). This is the non-seasonal humid zone of the Southeast Asian dipterocarp forests. Patches of rain forest, or outliers, are found in southern Thailand, in Sri Lanka, India, northern Queensland in Australia and on the Melanesian islands of the Pacific. Box 3 describes the tropical forest types in Malaysia, as an example of the complexity of tropical forests in general.

Box 3. The rain forest types of Malaysia (adapted from Symington 1943 and Wyatt-Smith 1964)

Malaysia's forests have been categorised into types that have been influenced by either climatic or edaphic factors. Climatic climax forest types traverse an altitudinal gradient, whereas the edaphic forest types are found in the lowlands.

Climatic climax forest	Edaphic climax forest
Lowland dipterocarp forest	Heath forest
Hill dipterocarp forest	Forest over limestone
Upper dipterocarp forest	Forest over ultramafic outcrops
Montane oak forest	Beach stand vegetation
Lower ericaceous forest	Mangrove forest
Montane subalpine vegetation	Brackish-water forest
	Peat swamp forest
	Fresh water swamp forest
	Seasonal swamp forest

The lowland, hill and upper dipterocarp forests are stratified into three storeys of trees. The Dipterocarpaceae is the main tree family in the forests of Southeast Asia and forms a high proportion of the upper strata of the forest. At least three-quarters of the forests of Southeast Asia are dipterocarp forests, and in Malaysia, they form over 86% of the forested areas. Dipterocarps do not grow at higher altitudes so are not found in the montane oak forest, which consists of two storeys of trees, or the ericaceous forest, which has a single storey of trees. The montane subalpine vegetation is low shrub vegetation on mountain peaks. The edaphic climax forest includes the low-lying swamp forests and forests on sites with extreme drainage and deficiency in available moisture due to violent winds. In Malaysia, swamp forests, including mangroves, form over 12% of the forested land.

Significant conversion of lowland forests to other land use began with the increase in tin mining activities in the western parts of Peninsular Malaysia in the middle of the nineteenth century and the beginning of rubber *(Hevea brasiliensis*) plantations at the start of the twentieth century and, later, the cultivation of oil palm (*Elaeis guineensis*), which was planted when rubber prices dropped sharply in the world markets in the 1960s. Forest conversion to cash crop agriculture, mainly oil palm, intensified from 1971 to 1990. Over the period 1970 to 1989, forested land in the whole of Malaysia was reduced by 23 per cent. (Manokaran 1992). Almost all the forests cleared were lowland forests, the largest reservoir of genetic variation of the dipterocarps and a major storehouse of biological diversity. Species richness in these forests is exemplified by the results of enumeration of woody trees of 1 cm dbh and greater in a 50-ha plot in Pasoh in Peninsular Malaysia. A total of 820 species in 294 genera and 78 families were recorded, this being almost one third of the total number of tree species found in Peninsular Malaysia, indicating that small areas include a surprising large percentage of a region's tree and shrub flora (Kochummen *et al.*, 1990).

Species diversity

Where seasonality of rainfall occurs, it produces a strong temporal effect on primary production (Orians *et* al., 1996). Productivity varies considerably among the primary tropical forest types; Lieth and Whitaker (1975) and Murphy (1975) provide the following data: tropical rain forest: 1800-3210 g/m^2; cloud forest: 2400 g/m^2; dry deciduous and mixed tropical forests: 1040-1230 g/m^2; for seasonal forest, a single estimate of 1340 g/m^2 from west Africa, and for mangroves: 930 g/m^2 from the Caribbean and 1000 g/m^2 at 10 to 25 years of age at Matang, in Peninsular Malaysia. These data show a primary productivity of 2–4 times that recorded in boreal forests and correlate broadly to a general latitudinal reduction in diversity of plants and animals north from the tropical forest biome.

Tropical forests are the most species rich and diverse forests on earth, estimated to contain at least 50% of all plant and animal species (Myers 1986). This is especially true for wet tropical forests, where, for example, some 700 tree species have been recorded in 10 selected 1-hectare plots in Borneo (UNEP, 1995). Within tropical moist forests, species richness varies greatly by region and some tropical moist forests actually have relatively low tree species diversity. In the Amazon Basin, for example, less than 90 tree species per hectare have been recorded in the eastern portions compared with nearly 300 species/ha in the western areas (WCMC, 2000). Mangroves have relatively low terrestrial species richness, with counts in some river deltas of about 30 species (IUCN, 2000), although the aquatic life they support is diverse and abundant. African rain forests have fewer plant species than other tropical regions (by about 20%), with several pantropical genera and families (e.g., Lauraceae, Myrtaceae and Palmae) being either absent or poorly represented (Jacobs 1981). Lianas and epiphytes are also less abundant in African rain forests compared to other tropical regions (Jacobs, 1981).

Few tropical genera are pantropical and endemism is much higher in this biome than in the temperate or boreal forest biomes (UNEP, 1995). For example, in fourteen areas with exceptionally high species richness in the tropics, on about 300,000 km^2., more than 37,000 plant species can be found (Myers, 1990). Tree species richness declines as altitude increases and as climate becomes more seasonal (Orians *et al.*, 1996). The mixture of many tree species, with few individuals of each, in a given forest area is a key feature of tropical forests and one which distinguishes them from forests in the boreal and temperate biomes. This feature is significantly related to a predominance of dioecious species and to a seed dispersal relationship with animals in the tropics, compared to boreal and temperate forests where wind is often the medium of seed dispersal (Orians *et al.*, 1996). Low density of individual species has particular consequences with respect to the necessity for large areas for preserving populations (Gentry, 1992). Where tropical forests with single dominants do occur (usually dry forest), there are no corresponding species among the regions. In the Americas, *Eperua* and *Mora* dominate such tropical forests, in Africa, *Gilbertiodendron* is a common dominant, dipterocarps dominate in areas of Southeast Asia, in Indo-Malaysia, *Agathis* is sometimes dominant, while in tropical Australia, *Eucalyptus* is the dominant genus in low-richness stands (Whitmore, 1990).

In rain forests, epiphytes, although common to all regions, are highly distinct and certain families predominate (Gentry, 1992) such as: Orchidacae in Africa; Orchidacae, Bromiliaceae and Cactacae in the Americas; and Orchidacae, Asclepiadaceae and Rubiaceae.[7] in Indo-Malaysia: Lianas are another important component of the structure of tropical rain forests, absent from the other biomes. They make up 8% of the species (in Borneo 150 genera exist) and are indicators of an undisturbed state of forests (Jacobs, 1981).

Twelve genera and some 470 species of the family Dipterocarpaceae are found in the rain forests of the Indo-Malayan region, ranging from the Seychelles through Sri Lanka to the south of Peninsular India, east to India, Bangladesh, Myanmar, Thailand, Indo-China, to continental South China (Yunnan, Kwangsi, South Kwangtung, Hainan) and through Melanesia (natural botanical kingdom comprising Peninsular Malaysia, Sumatra, Java, Lesser Sunda Islands, Borneo, the Phillippines, Celebes, the Moluccas, New Guinea and the Solomons) (Ashton, 1982). With the exception perhaps of New Guinea and the eastern part of the region, the tropical rain forests of the Indo-Malasian region are characterised by family dominance of the Dipterocarpaceae (See Box 1).

With respect to fauna, the forests of South America and Asia maintain very high animal species richness compared to the African tropical forests (UNEP, 1995). The rivers of the Amazon Basin host the most diverse fish population in the world and the insect populations present in its canopy also have high species richness (WCMC, 1999). Wilson (1986) recorded 43 species of ants, belonging to 26 genera, on a single tree in Peru, about the same number of species as the entire ant fauna of the British Isles. It is not unusual for a square kilometre of forest in Central or South America to contain several hundred species of birds and many thousands of species of butterflies, beetles and other insects (Wilson, 1986). Stattersfield *et al.* (1998) note that, of the total world forest avifauna, 88% are endemic to tropical forests, and of those, more than half are found in wet forest types.

Tropical dry forests generally host a lower species richness, with fewer endemics than tropical moist forests, although still significantly higher than in temperate forests. The richest dry forests, found in northeast Mexico and southeast Bolivia, have an average of 90 tree species per hectare (WCMC, 2000). Dry forests are more similar in species richness to their moist counterparts in terms of mammal and insect species. Tropical dry forests are noted for their highly endemic mammal populations, especially insectivores and rodents.

An important feature of cloud forests and some other montane forests lies in their high species richness of epiphytes, shrubs, herbs and ferns (Gentry, 1992). These species increase with altitude in the humid tropics whereas in the warmer, lowland tropical forest types, they tend to be less frequent. In addition, cloud forests often contain high numbers of rare endemic plant and animal species or subspecies, such as the mountain gorilla (*Gorilla gorilla beringei*) in Central/East Africa, and the quetzal (*Pharamachrus mocinno*) of Central America (IUCN, 1995). The percentage of endemic species is even higher in cloud forests on island mountains, such as those in Hawaii and in the French overseas territories of Reunion Island and New Caledonia.

Mangroves may form very extensive and productive forests. Throughout the tropics, there are about 60 species of trees and shrubs that are exclusive to the mangrove habitat, the important genera being *Avicennia, Bruguiera, Rhizophora, Sonneratia* and *Xylocarpus*. There are also important, non-exclusive species associated with the mangroves, including the fern *Acrostichum* spp., and trees such as *Barringtonia racemosa*, *Hibiscus* spp. and *Thespesia* species.

High species richness in the tropical biome may be the result of the large range of available microhabitats and niches, the absence of mountain systems or their north-south orientation permitting ease of migration and a lengthy period without major disturbance (e.g. glaciation) (UNEP, 1995). High productivity is

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[7] Not all Bromeliaceae are epiphytes; Cactaceae, Rubiaceae and Asclepiadaceae are not primarily epiphytes.

sustained annually, as opposed to seasonally, in many tropical areas which allows multiple breeding seasons and results in less movement away from a home range to avoid seasonality (Margalef, 1968; Richards, 1969). Further, in places such as Madagascar and the large number of tropical island habitats in Southeast Asia and the Caribbean, a high level of endemism is found because of their isolation (Margalef, 1968).

Plantations in the tropical biome

In the tropical forest biome, plantations have been reported to be a great success in terms of rapid growth due to favourable conditions, a fact that has motivated the adoption of plantations as a way to improve local forest productivity. Given the high biodiversity present in tropical ecosystems, plantations are particularly vulnerable to diseases and pests, especially when exotic species are involved. In spite of this, plantations have been promoted as manufactures have been increasingly accepting of plantation wood fibre, especially *Eucalyptus*. As a result, a great number of large pulpwood plantations have been established in Indonesia, South Africa and Chile and the area of plantations is still increasing in other countries such as Malaysia, Vietnam, Thailand, Uruguay, Paraguay, Argentina, Venezuela, Colombia, Mexico, Congo and Swaziland (WRM, 1999).

Several examples of plantations in tropical forests can be cited. In the Amazon region, Brazil has experienced a number of ambitious projects such as the Jari Project and the Projeto Grande Carajás, Aracruz celulose pulp plantation (Rankin, 1985; National Geographic, 1990; World Resource Institute, 1998). In Congo, the eucalyptus plantations at Pointe Noire have been successful as productivity has improved due to an imaginative and dedicated tree breeding project. On the other hand, in Senegal and in Java, Indonesia, where there are about 600,000 ha of teak (*Tectonia grandis*), there is some concern for site deterioration due to repeated planting, soil erosion and loss of organic matter (Evans, 1999), especially as grand scale plantations are always more vulnerable to natural disasters.

Agro-forestry

Agro-forestry can be considered to be a productive system which conserves biodiversity, mainly when it is based upon traditional knowledge, as it causes minimal land degradation and preserves biodiversity as well as native crop diversity (Toledo, 1980). It has been successfully used as a sustainable forest management practice in some parts of Central and Latin America. For instance, in Mexico and Columbia, maize cultivation is combined with rows of *Swietenia macrophylla*, *Cedrela mexicana* and *Cordia alliodora*, while *Erythina* and *Cordia* shade coffee and cocoa plantations (Burniske, 1993). At present, Costa Rica is developing an ambitious programme that combines scientific research with multiple-use agro-forestry projects that include fruit-tree and forest cultivation, with the social support of landowners. These projects aim to provide several environmental services that refer to carbon sequestration and biodiversity conservation [see e.g. CREP].

C. SCIENTIFIC CONSIDERATIONS RELATING TO FOREST BIOLOGICAL DIVERSITY

Forest biological diversity is quantified at several scales by assessing the genetic diversity within species, counting the number of species per unit area (local, regional, national, continental, global), determining areas and types of forest ecosystems, determining communities of species associated with forest ecosystems, determining numbers and arrangement of forest types and ages and through describing landscape structure. A recent shift in forest management methods in many countries, to focus on 'ecosystem management', is a result of the recognition of the need for a scaled-up approach. This change accompanied the further understanding that most species within systems cannot be monitored sufficiently to ensure independent long-term viability. Monitoring programmes are an important component of the sustainable management of forests to ensure that biological diversity is maintained in time and over space.

I. Genetic diversity

Genetic diversity of a given species can be assessed by different techniques, including studies of morphological and metric characters, use of biochemical and molecular markers and observations of ecogeographic variation. A wide range of powerful molecular markers (DNA and isozyme) is now available for assessing genetic diversity in particular organisms (e.g. Hamrick and Godt, 1990; Szmidt, 1995; Gillet 1999). Depending on the marker used, the information from such studies may provide different insights into the level and structure of genetic variation and the evolutionary processes associated with species development and maintenance. For example, in *Acacia mangium*, virtually no variation was detected using allozymes (Moran *et al.*, 1989); however an RFLP study revealed considerable within- and between-population variation with greater variation being correlated with better field performance (Butcher *et al.*, 1998). More recently, microsatellite markers have demonstrated much higher variability than RFLP markers in the same species, permitting greater genetic discrimination between individuals. Microsatellite markers will be of great utility in mating system studies (Butcher *et al.*, 2000). Increasingly, attention is being given to ensure that the limited resources for genetic research in rare species are used for studies which will provide information of direct use in developing conservation management plans for the species (e.g., Hogbin *et al.*, 2000). Quantitative and adaptive traits, e.g. physiological tolerances to various abiotic stresses and disease resistance, can be assessed through a variety of field and laboratory techniques.

Genetic diversity is high in widely distributed forest-dwelling organisms with a sexual reproductive system, and will mirror environmental variation. However, with the exception of some species of plants, larger mammals and birds, there is limited information on the genetic diversity present in most forest-dwelling species. For plants, the level of genetic variation, as assessed through isozyme studies, has been found to vary with factors such as geographic range, breeding and pollination systems and seed dispersal mode (see Table 4).

Table 4: Levels of allozyme variation in plant species (from Hamrick and Godt 1990)

CATEGORY	Polymorphic loci (%)	Number of alleles per locus	Index of genetic diversity (H_e)
GEOGRAPHIC RANGE			
Narrow	45	1.8	0.14
Regional	53	1.9	0.15
Widespread	59	2.3	0.20
BREEDING SYSTEM & POLLINATION			
Selfing	42	1.7	0.12
Mixed mating system - animal pollinated	40	1.7	0.12
Mixed mating system - wind pollinated	74	2.2	0.19
Outcrossing – animal pollinated	50	2.0	0.17
Outcrossing – wind pollinated	66	2.4	0.16
SEED DISPERSAL			
Gravity	46	1.8	0.14
Animal – ingested	46	1.7	0.18
Wind	55	2.1	0.14
Animal – attached	69	3.0	0.20

Forest tree species are typically long-lived, outbreeding and generally highly heterozygous organisms that have developed natural mechanisms to maintain intra-specific variation (see e.g., Palmberg-Lerche, 1993). These mechanisms, combined with their often variable natural environments, both in time and space, have contributed to forest tree species evolving into the most genetically variable of all living organisms (Libby, 1987). This genetic diversity should be recognized as a major and vital component of forest biological diversity.

The genetic diversity present in forest species other than plants and high-profile vertebrates has received little study. Genetic studies carried out on organisms other than plants often address the extent and temporal or spatial patterns of genetic diversity and their links with a number of environmental or historical factors. Although not specific to forest ecosystems, several good data sets are now available to trace the influence of global climate changes, especially since the Quaternary ice ages, on genetic diversity of species and populations in boreal, temperate and tropical zones (Hewitt, 2000). Some birds, insects or amphibians, sensitive to alterations in forest cover, structure or composition, or heavily dependant on a single tree species, have been tested for use as indicators of the degree of forest alteration or degradation. The mean expected heterozygosity in allozyme data for insects is 0.137 + 0.004, for mammals is 0.067 + 0.005, birds is 0.068 + 0.005, and amphibians 0.109 + 0.006 (UNEP, 1995). Such low heterozygosity can result in high rates of allele fixing should normal dispersal rates be reduced through fragmentation of forest habitats. Groups with species that fly, enabling high rates of gene flow, have shown highest levels of heterozygosity

Theoretical population genetics provides a background for the conservation of species, populations and genetic resources (Delden, 1992). The use of models to determine effective population size and rates of genetic drift has led to estimates of the minimum population size needed for survival of a species over various specified periods of time. Understanding of population genetics has led to the conclusion that maintaining several distinct populations of a species increases the overall effective population size, thereby increasing the survival potential for species with small populations (Nunney and Campbell, 1993). Genetic diversity is also the essence of natural selection. Robustness within populations is dependent on a high level of genetic variability, which enables individuals within populations to survive environmental variation. From an anthropocentric perspective, genetic diversity can be viewed as a resource, which has the potential to provide new drugs, better crops and improved resistance to pathogens among forest trees.

Genetic variation may be associated with allelic variation in the DNA present in the nuclear and mitochondrial genomes as well as with chromosomal variation (ploidy levels). In plants, genetic variation may also be due to differences in chloroplast DNA. At the genetic level, individual alleles are usually quite discrete and recognisable. However alleles cannot be identified without expensive and relatively sophisticated detection systems, usually relying on gel electrophoresis of molecular markers. Furthermore, it is not simply numbers of alleles, but how they combine in forming multilocus genotypes that determines effective genetic diversity. The huge number of gene loci also makes it impossible to quantify more than a small sample of total genetic diversity for a given species/population. Gene diversity as measured by expected heterozygosity (H_e) or other parameters, represents only one type of genetic variation. Many adaptive traits are polygenic, i.e., controlled by the combined effects of many loci. Such phenotypic traits have a continuous distribution, often referred to as quantitative variation. It is unclear how to relate genetic variation at the molecular level with the quantitative variation observed for polygenic traits. Direct characterisation of individual genes is not possible for quantitative traits, but recently developed statistical techniques allow inference from estimation of genetic variances (Hamrick and Godt, 1990).

High-profile vertebrates studied for their within-species diversity may receive high media coverage and some of them have become emblematic species for nature conservation campaigns and recovery efforts. Genetic diversity has been examined for some endangered vertebrates, where populations have become so small that inbreeding depression became a concern. There has been considerable debate over the role of genetics in population declines. In some cases, declines that were previously thought to have resulted from a loss of heterozygosity may have been the result of other factors (e.g., cheetahs - Caro and Laurenson, 1994). Inbreeding, however, can certainly be a proximate cause of decline, and in very small populations the genetic consequences of inbreeding can become a paramount problem.

The value of the genetic variation in economically important fish species living in flooded forests (e.g. seed- and fruit-eating tambaqui *Colossoma macropomum* in the Amazon, trey riel *Henichrhynchus siamensis* in Tonle Sap, Mekong Basin) is just starting to be recognized in aquaculture development. Genetic diversity of evertebrates or micro-organisms restricted to forest habitat is a nearly complete *terra incognita*, with the exception of a few species of mycorrhizal fungi or rare vertebrates.

While some large-scale forest tree inventories support other studies on epiphytic plants, arthropods, fungi and grasses (e.g. inventories of the Pasok Forest Reserve in Peninsular Malaysia), these side studies do not generally address the variation within-species. There has so far been no comprehensive compilation of the information on all biological diversity available at within-species level in forest areas. Since genetic-level assessment of biological diversity is generally limited to a few species, a large degree of genetic diversity alteration is probably unnoticed in disturbed forest ecosystems. Lleras *et al.* (1992) have shown that, while human-induced species extension in the Amazon Basin has not been extensive to date, substantial loss of genetic variability has occurred in many species important to humans, both in the Basin and in the ecotone, between the Amazon and the cerrado.

II. Species diversity

Species, or alpha diversity, is a measure of the number of species of all or various taxa per unit area, properly referred to as species richness. Another measure is gamma diversity, which reports the sum of alpha diversities across a broader region. Knowledge of the diversity of plant and animal species found in forests and their distribution is incomplete. While some groups, such as mammals and birds are often documented reasonably fully, other taxa such as invertebrates and microbes remain virtually unknown (FAO, 1999). The UNEP Global Biodiversity Assessment (UNEP, 1995) suggested that of the estimated 13.6 million species of organisms worldwide (including aquatic and oceanic species), less than 1.8 million have been described. The majority of species are insects (8 million), fungi (1.5 million) and bacteria (1 million) and the vast majority of these are undescribed. Even among terrestrial plants, many species are incompletely known in some regions. For example, only 30-40% of 15,000 botanical specimens held in the Royal Forest Herbarium could be identified during the Flora of Thailand Project (OEPP, 1997). For the poorly known groups of organisms, the estimated number of total species often entails considerable, rather tenuous extrapolation. For example, the commonly quoted estimate of global fungus diversity of 1.5 million species (Hawksworth, 1991) has relied on extrapolation from studies in temperate zones. However, a recent study of fungi growing on the tropical palm, *Licuala*, suggests that even 1.5 million may be a conservative estimate (Fröhlich and Hyde, 1999). Further, even when a species is identified, considerable work remains to determine its range, habitat requirements and relative priority for conservation.

It is well known that primary tropical forests, particularly tropical wet forests with their high per unit area productivity, support much higher species richness than either temperate or boreal forests (UNEP 1995). Latitudinal gradients in species richness have often been reported (e.g., Stevens 1989, Ricketts *et al.*, 1999). However, the fundamental processes that influence forest biological diversity are more or less common to all forest biomes. Species richness in forests is affected by several factors operating at local and regional scales, including long-term history (e.g., glacial events), ecological space available for niches, forest productivity (which is ultimately related to climate, soils and available water), extent of forest cover, forest landscape heterogeneity, species interactions and, for vertebrates, body size (Holling, 1992; Ritchie and Olff, 1999). At large scales, Thompson (2000) showed a direct linear relationship between bird, mammal and salamander/reptile species richness and net primary productivity in boreal and temperate forests in Canada. Helle and Niemi (1996) reported that numbers and species richness of birds increased with stand age-class in boreal forests, although the proportion of Neotropical migrant species declined with successional age in the Palaearctic region. Some longitudinal variation also exists in the species richness of mammalian herbivores of the boreal region (Danell *et al.*, 1994). At local scales in primary forests, ecological niche space may be very high because of the complex vertical and horizontal structures of this forest type. Patches of forest have differing suitability as habitat for a given species depending on their size, presence of required structures, plant species composition, age and isolation from other patches; these factors influence patch quality. Some patches may be occupied by a given species for extended periods, while others may be unoccupied or occupied infrequently. Quality of a patch to a particular species may be affected by its isolation from other forests because the species is incapable of migrating across the hostile surrounding areas (Lovejoy *et al.*, 1986). Hence, not all available habitat is necessarily occupied. For this reason, landscape structure and connectivity among forest patch size and continuity are extremely important in maintaining biological diversity (Hanski, 1999a, b).

Species/unit area curves offer a quantified prediction that the larger the area of forest, to some upper size limit, the greater the number of species that can occupy the forest (Preston, 1962). The asymptote of the curve depends to a large extent on beta diversity of the landscape. A high beta diversity will result in high regional diversity, but with low local richness or alpha diversity (Cornell and Lawton, 1992). The relationship between species richness and forest area is valuable for predicting numbers of species in a sample area and possible changes as a result of forest management. The same relationship also predicts that most species in small patches will be generalists, and that specialist species, with their stricter habitat requirements, will primarily exist in self-sustaining populations in large continuous forest areas. This is particularly true of animal species with large body sizes and/or large home range requirements, such as most carnivores. However, as forest patches become smaller and the distance between them becomes larger with treeless areas in between, populations of smaller-bodied species, such as invertebrates, also become affected. Further, most species that have narrow habitat tolerances are area-sensitive. Thresholds exist in the reduction of continuous forest into patches beyond which species extinctions occur in a rapid and non-linear manner that is often unpredictable (Andren, 1994; Haila, 1999; Hanski, 2000). Andren

(1994) indicated that when forests are reduced to 10-30% of the original cover, species' declines become abruptly non-linear.

Keystone species

Keystone species are those whose removal from an ecosystem would have a disproportionately large impact on processes in that system (Power *et al.,* 1996). Both plants and animals can be keystone species. For example, in Peru, twelve species of figs and palms maintain all frugivorous animals for three months each year (Terbourgh, 1986). There have, indeed, been instances of frugivores declining in association with loss of their food trees in logged tropical forests (Frumhoff, 1995). It is the degree of interaction, or the linkages, between a particular keystone species and other species within the system that is important. Keystone species do not always occupy an elevated trophic status (Power *et al.,* 1996), and it is likely that most keystone species have not been identified as such because many are soil invertebrates, pollinators, mutualists or pathogenic organisms (Krebs, 1985; Power *et al.,* 1996). Jones *et al.* (1994) used the term 'ecosystem engineers' (or keystone modifiers, after Mills *et al.* [1993]) to refer to species that create, modify or maintain certain environments. These are keystone species because they play important functional roles in ecosystems by creating habitats required by other species. Therefore, the removal of the engineering species from an environment results in species impoverishment beyond loss of the species itself. In North America, beavers (*Castor canadensis*) are a keystone species because they modify several hectares of forest by creating impoundments through damming streams. These impoundments are then used by many aquatic and semi-aquatic species. Ants may be other keystone species as they are responsible for seed and spore dispersal of certain plants (including many fungi), soil mixing and aeration and decomposition of wood. Ants can also have specific competitive effects on the distribution of other ant species and ground-dwelling arthropods, certain canopy arthropods and some vertebrates, including woodpeckers and ant eaters in tropical forest systems (Puntilla *et al.,* 1994; Elmes, 1992; Holldobler and Wilson, 1990). Some woodpecker species may also be considered keystone modifiers because they excavate cavities in large trees that are subsequently essential for successful reproduction of many other species. If the habitat requirements of woodpeckers are provided through careful planning, then habitat will be created for secondary cavity-using species including forest bats, cavity-nesting passerines, cavity-nesting owls, certain wasp species and tree squirrels (Thompson and Angelstam, 1999).

Species that concentrate their populations annually

An important aspect of the life history of many animal species is their movement to restricted areas that meet their needs for some portion of the year. During periods when a species congregates at high density in localized areas, a large proportion of local populations, or even the entire population, is particularly susceptible to disturbance or habitat loss. In forest ecosystems, there are numerous examples of species that move into localized high-density areas, often in winter. For instance, many species of amphibians congregate to breed in vernal pools in forests or forests margins. An example of a migratory species that concentrates is the monarch butterfly (*Danaus plexippus*) of North America (Malcolm and Zalucki, 1993), which breeds over much of the United States and Canada and overwinters in small areas in Mexico. A component of ecosystem management is therefore to safeguard the important or key habitats associated with species that concentrate and such species are arguably a special case for protection to maintain biological diversity (Thompson and Angelstam, 1999).

Endangered species

Across all biomes, the *2000 IUCN Red List of Threatened Species* lists over 11,000 species threatened[8] with extinction (although this is based on a complete sample of only mammals, birds and coniferous trees, and very incomplete assessments of other taxonomic groups). While this is less than one percent of the world's described species, it includes 24% of all mammal species and 12% of all bird species. While

[8] On the meaning of "threatened" see the 1994 IUCN Red List Categories and Criteria. IUCN. 1994. *IUCN Red List Categories.* Prepared by the IUCN Species Survival Commission. IUCN, Gland, Switzerland. Available at: http://iucn.org/themes/ssc/redlists/ssc-rl-c.htm

plants have not yet been evaluated systematically, in an earlier work Walter and Gillett (1998) found that at least 12.5% of flowering plants were threatened. The 2000 Red List also estimates that 25% of reptiles, 20% of amphibians and 30% of fish species are threatened. It has been estimated that current species extinction rates are between 100 and 1,000 times higher than the natural background rate and before humans began to cause extinctions (Pimm *et al.,* 1995; Lawton and May, 1995).

It is difficult to provide comprehensive estimates of how many forest species are threatened since there is no globally accepted habitat classification system. More specific data are available for individual tree species and some forest habitat types. For example, 900 threatened bird species rely on tropical rainforests, with 42% of these occurring almost exclusively in lowland rain forest and 35% occurring in montane rain forest. Among mammals, 33% of those threatened occur in lowland rain forest and 22% in montane forest, though it is not clear what proportion of these mammals are totally dependant on forests for their survival (Hilton-Taylor, 2000). *The World List of Threatened Trees* (Oldfield *et al.*, 1998) documents over 7,300 species facing extinction (Table 5). Based on gross estimates of the total number of tree species in the world, this represents about 9% of the world's tree flora. Although not specifically targeting endangered or threatened tree species because of their endangered condition, the list of priority species established by the FAO Panel of Experts on Forest Gene Resources (FAO, 2000c) also suggests that around 9% (48 species out of 524 tree species recorded) of the world's most important trees are threatened, at species or population level.

Table 5: Globally Threatened Tree Species[9]

1994 IUCN Threat Category	**Number of Tree species**
Extinct	77
Extinct in the Wild	18
Critically Endangered	976
Endangered	1,319
Vulnerable	3,609
Lower Risk: near threatened	752
Lower Risk: conservation dependent	262
Data Deficient	375
Total	**7,388**

Species and communities as indicators of forest change and habitat loss

Indicator species are generally readily recognizable and are usually discrete biological entities whose predicament and/or conservation needs easily capture the imagination of the general public and government agencies alike. It is worth noting that tree species diversity (provided that this can be assessed) may prove to be an adequate surrogate for overall species diversity in most forest ecosystems. Trees as indicators may be particularly relevant for those forest ecosystems where many animal and fungal species are obligate associates of particular tree species (e.g. Campbell, 1989). Comparatively good data are available on the distribution of many tree species (Anon., 1996; Van Bueren and Duivendoorn, 1996). Indicators may be used to suggest the effects of change within a system at particular scales, or to indicate population trends that result from altered ecological processes. Most often indicators are chosen as 'umbrella species', defined as indicator species that correlate strongly with the presence of other species**,** (i.e. their presence indicates, with very high probability, the presence of several to many other species). All umbrella species are indicators, but not all indicators are necessarily umbrella species.

[9] Adapted from: Table 2: Summary of the number of tree species assessed according to the 1994 IUCN threat categories for inclusion in *The World List of Threatened Trees.* (Oldfield, 1998, p. 17.)

The use of indicator species monitoring is an integral component of a programme of ecosystem management.

In similar forest patches, not all subsets of species are necessarily the same, and across landscapes, habitats are not uniform. These variables can lead to a wide disparity in species richness measures across samples from a landscape. Community assemblage may depend to a large degree on history and stochasticity, as well as availability of suitable habitats, environmental gradients and the supply of species through immigration. Numerous processes affect a species' capacity to occupy a forest stand, including competition for resources and demographic factors. In studying the effects of change in forested systems, the concept of alternative states in the same forest types must be considered. In other words, the presence or absence of a species in an area may be simply a manifestation of the same community in a different state, with no change in its functional capacity. Further, under the metapopulation theory, at any given time a patch of habitat may or may not be occupied by a certain species (Levins, 1992). Therefore, from a forest management perspective, the lack of a particular species in an area is not necessarily cause for concern under a properly replicated census design. On the other hand, absence of that particular species may indeed suggest serious problems. Efforts to map species diversity have been limited, mainly to regional levels, and have not usually included a separate analysis of forest species. Because of the difficulty and vast resources needed to make an inventory of the total number of species in an area, certain better-known genera or families are often used as surrogates for the presence of other species or to indicate functioning ecosystems. However, species may respond individually to change, as predicted by the unique niche hypothesis, suggesting that changes in numbers of one species do not necessarily correlate to changes in others [Reference needed]. Care must, therefore, be taken in the selection of indicator species, as well as in interpretation of census results. The characteristics of useful biological diversity indicator species are discussed in Caro and O'Doherty (1998). Hunter (1990) suggested that indicators were species that should be monitored to reveal the presence/absence of certain specific conditions or structures (fine filters) that were missed with coarse filter (or forest type) management. Indicators may be employed to assess change in a long list of forest aspects including processes and structures from sites to landscapes, for example, landscape heterogeneity, or site diversity.

There is a substantial body of literature on community responses as indicators of pollution effects (e.g., Cairns, 1985). If forest processes have been altered, then the structure of animal communities will likely also have been affected (Holling, 1992). It often makes sense to examine species assemblages as indicators, rather than try to monitor individual species, in part because of a narrow focus in following individual species and also because of the general lack of autecological information (Kremen, 1992; Dufrêne and Legendre, 1997). For many taxa of animals or plants, all (or most) species in an area may be sampled simultaneously. Discarding the broader data set in favour of a few chosen species is not cost-effective when the tools are available to use all the data. For example, theoretically all songbirds can be recorded in singing male point counts, all salamanders may be equally vulnerable to pitfall trapping or shelter-board surveys, and most small mammals species are readily captured in traps. Therefore, many of these data sets lend themselves to community analyses. When choosing the communities or species assemblages to monitor, their sensitivity to forest change at multiple scales relevant to the problem should be considered. Westoby (1998) noted that it is important to choose the appropriate scale for assessing local species richness and, because the analysis is statistical, an adequate number of points with which to perform regressions must be obtained. Ordination of communities can aid in decisions about what communities might perform well as indicators, or the properties of a particular group of species as indicators at a given scale. The pattern of co-variation over time among species within a community may indicate impacts of effects.

Population viability analysis

The presence of a species does not necessarily indicate that the population is sufficiently large to ensure long-term survival of the species. Further, even large population size does not guarantee population survival (Mangel and Tier, 1994), hence populations of species in more than one location are important to long-term persistence. Any monitoring programme for rare species should be concerned with population viability analysis (PVA) and also modelling to determine minimum viable populations (MVP) for key or rare species. The outputs from such modelling suggest critical population sizes needed to maintain genetic diversity. PVA modelling considers three main points: genetic factors, population demography and ecological factors (Boyce, 1992). Among the important ecological factors are the occurrence of ecological thresholds such as those caused by fragmentation and Allee effects (inability to locate mates) due to habitat loss and change in habitat structure. More recently, Hanski *et al.* (1996) have developed the term minimum viable metapopulation, to describe the common situation of limited emigration from source areas, such as protected areas, to surrounding habitats. They have suggested that small numbers of a local population have a high probability of extinction due to stochastic events. However, such modelling requires certain minimal data, which are most often not available, including means and variances of vital population demographic statistics and how these may differ in various habitat patches, dispersal rates, dispersal distances and success (Doak and Mills, 1994). Thus, unless considerable research is undertaken, a detailed understanding of the likelihood of survivorship in a population will not be obtained and yet this may be crucial to management decisions.

Selection of elements for an inventory and to monitor, and monitoring methods

An inventory is a compilation of data, often with maps, on the aspects of biological diversity chosen to monitor, while monitoring refers to the collection of data over time. Forest biological diversity is an issue of scale. Therefore, compiling an inventory and monitoring of biological diversity is a hierarchical procedure, requiring various techniques and technologies. Depending on technical capacity and state of knowledge, aspects to monitor, ranked from least to most demanding, include: forest types (area), forest ecosystems (area, age classes), landscape structure (patch sizes, associated selected variables), species (indicators, endangered species, key species, useful species) and genetic diversity. A starting point for any monitoring programme is an understanding of the distribution of the element to be monitored.

A serious limitation to detecting change in global forest conditions is the lack of capacity in many countries to perform even large-scale monitoring of forest types and landscape structure and, more generally, an incomplete knowledge of forest species and forest ecosystems. Classification systems for each level of biological diversity must exist as a pre-condition to compilation of an inventory and monitoring

III. Ecosystem and landscape diversity

An ecosystem concerns scale relative to the organism or group of organisms or even the ‘question’ being considered. For forests, an ecosystem generally refers to a clearly identifiable and distinct area of certain species and functional and abiotic components. Ecosystem function is the total of all processes that occur within the ecosystem including production, nutrient cycling and water movement. Many forest ecosystem classifications exist, usually based on an ordination of soils and moisture regimes, which combine to support a particular kind of forests, level of functions and associated organisms. The role that species may play in ecosystems was briefly discussed above; considerably more information is needed with respect to this important concept (see Chapter II). Several hypotheses have been advanced to explain the role of species in ecosystems. These include (Vitousek and Hooper, 1993; Lawton, 1994): a) a null hypothesis, where ecosystem function does not change with gain or loss of species; b) the idiosyncratic hypothesis, which predicts that there will be a change in function related to species loss or gain, but the magnitude and direction may not be predictable; c) the rivet hypothesis (Ehrlich and Ehrlich, 1981), which predicts that all species contribute to function and that loss/gain in function is equitable for each, or that thresholds in numbers of species exist that once crossed, a disproportionate amount of function is lost; d) the redundant species hypothesis that predicts only some species contribute to function and the roles of all others are redundant (Walker, 1992; Lawton and Brown, 1993). There is little evidence to support any of these hypotheses, particularly for vertebrate species, and certainly we generally do not know the form or shape of the relationship. As a general principle however, theory and experimentation suggest that complex ecosystems with large leaf areas are more productive and maintain more niche space, and hence more species, than less productive ecosystems with minimal richness in primary producers (Tilman *et al.*, 1996, 1997). However, rich systems are also likely to maintain a high species redundancy (Lawton, 1994). Although Sugihara (1980) suggested communities form around a hierarchical niche structure ('sequential breakage model') that obviates any redundancy, work by Tilman and Downing (1994) revealed that reducing the number of primary producers from 25 to 10 did not alter net primary productivity in a field experiment. Opposite results were achieved by Symstad *et al.* (1998) who showed a decline in plant biomass with species reduction in grassland systems. They also found that nitrogen retention (a measure of productivity) changed unpredictably depending on the species removed, supporting the idiosyncratic hypothesis. A further unknown is the role that rare species may play, under extreme events, in buffering an ecosystem against change (Schulze and Mooney, 1993). The relationship between stability in systems and species diversity and richness is still obscure and is a source of concern with respect to the use of forests.

Concerns over resource use and effects on biodiversity relate to questions about ecosystem resiliency. In other words, can these systems be used in a manner that permits them to return to a state that is similar to the state existing prior to harvesting and, hence, maintain the same species diversity (sustainable use)? If not, then they may return to an altered state, with little or no stability, which cannot support the biological diversity that existed prior to use. Because forests take many years to grow, this question of sustainability remains open. It is likely that using ecosystems results in less than optimal production in the system and that constant use impairs maximum production by altering the ecosystem state beyond certain thresholds. However, predicting thresholds, and hence levels of sustainable use, is problematic.

A second set of questions with regards to stability in ecosystems pertains to the functional roles that species may play within the system that are not directly related to productivity. For example, numerous species are responsible for pollination of plants, the primary producers. Without these species, which live off excess energy in the system, the system could not exist. This role may be a 'keystone' role as referred to above. Often, exotic (or non-native) species may affect stability in ecosystems by altering processes such as these.

Landscape is one of several scales of forest biological diversity (Noss, 1990), although landscape by itself is a concept of scale. As with ecosystems, a particular landscape is defined relative to the organisms, populations or processes under discussion. In the case of a forest landscape, the size reflects the broad-scale processes that cause variation in forest ecosystems. Processes that affect forest community structure are largely historical and operate at large-scales (Cornell and Lawton, 1992). An understanding of how past forest landscapes were formed and how processes have changed through history is needed to provide a perspective on new landscape structure that is inevitably altered by humans. Contemporary forestry and land-use practices can substantially alter forest landscape structure, often irrevocably, through land-clearing, changes to patch size and distribution, through losses of fine-scale structures and microhabitats (Haila *et al.*, 1994), away from heterogeneity towards homogeneity (e.g., Pickett and White, 1985), and through the creation of dynamics that differ from those under more natural disturbance regimes (Hunter, 1990; Virkkala and Toivonen, 1999). Many animal species, particularly those with large body sizes, respond to forest landscape structure. In addition, the effects of forest fragmentation on species, discussed above, operate at the landscape scale. Therefore, the distribution of forests across the landscape has implications for maintaining biological diversity.

Suffling (1991, 1995) has advanced the theory that landscapes are in "constant disequilibrium", in other words, stability at the scale of forest landscape never occurs. This view is shared by Baker (1995) on the basis of considerable modelling of northern temperate mixed forests of the United States. According to this theory, landscapes are constantly "catching up" with disturbance regimes that change at two temporal scales, the longer one in the order of several hundreds of years. Current climate data can not provide the basis for predicting that present northern temperate and boreal forest ecosystems, formed following disturbances 100-500 years ago, will recur at any particular point in the future, because forest response to disturbance is a complex process involving numerous interacting factors and time lags (Drake, 1990; Bonan, 1992).

The ultimate factors that control landscape structure act in a "top-down" manner, affecting ecological processes over large areas and long time scales, while more proximate factors influence individual patch sizes and their rates of change over shorter time periods. The local distribution of forest ecosystems is influenced by site conditions, as modified by disturbance events. The development of forest vegetation across a landscape is influenced by three ultimate factors acting at large spatial and temporal scales: physical factors (including soils, lithology, elevation and relief), climate (rainfall and temperature) and anthropogenic factors (including history and livelihood). Physical geography and climate set fundamental limits on forest development, while history represents the net (or 'combined' or 'cumulative') effect of specific events of forest growth, change and destruction over time. Coupled with the ultimate factors are proximate factors, those that act at more limited spatial and temporal scales to influence stand development.

Palaeoecological research implicates climate as a key factor in temperate and boreal forest change during the post-glacial period (Webb, 1986; Ritchie, 1987) and suggests that plant migration can generally track climate change, regardless of the speed of that change (Pitelka *et al.*, 1997). Payette (1992) notes that although climate and ecosystem processes can explain the migration and retraction of forest types in boreal and temperate biomes, competition is also an important factor in structuring plant communities. Forest development at large scales not only responds to climate, disturbance and forest management, but also to the ability of individual species to compete, to given specific historical sequences and stochastic events and to specific soil conditions.

At large scales, tree species richness correlates with many climatic variables, such as number of growing-degree-days, mean annual temperature, mean annual insolation and total annual precipitation. Net primary productivity is a variable that integrates many aspects of climate, such as length of the growing season, soil temperature and moisture regimes. A study testing this 'climate affects species richness' hypothesis for Ontario, Canada (an area of 1 x 10^6 km^2, over 13^o of latitude) revealed a highly significant positive relationship between tree species richness and net primary productivity (Thompson, 2000).

The association between topography, soils and forest types (or individual ecosystems) has been reasonably well studied (e.g. Sucachev, 1928; Cajander ,1948; Daubenmire, 1968) and it has been shown that topography and soil types exert strong influences on forest development. Relationships exist among slope, soil type, microclimate and soil moisture content, all of which can be used to infer local productivity (Brady, 1984). Graumlich and Davis (1993) reported that substrate governs the distribution of pines (*Pinus* spp.) and birches (*Betula* spp.) in the North American Great Lakes Basin over an area of hundreds of square kilometres. Knowledge of the associations between soil types and plants can be used to predict and model forest landscape structure.

Proximate factors, as stated earlier, are those that influence the development of forest vegetation at a smaller scale. The principal types of proximate factors are fires, wind, insect infestation, diseases, flooding and human interventions, as well as periodic changes in weather, competition among species and grazing by herbivores. At intermediate scales, human intervention is important to the structure of forest landscapes. However, it is doubtful that the complex inter-relationships of natural systems can be replicated by management (Hansen *et al.,* 1991; Hunter, 1993). Although one of the goals of current forestry practices in many jurisdictions is to emulate natural processes, the concept that fire or hurricane effects can be emulated through logging practices - a cornerstone of sustainable forest management - is unlikely to be valid. Forest harvesting, coupled with fire suppression, has altered forest stand composition. Reforestation strategies and policies have changed over time, as have silvicultural practices. These changes have led to the redirection of forest successional trends, particularly compared to natural pathways, and ultimately to changes in landscape pattern. An important factor affecting the distribution of individual species, and ultimately the landscape pattern, is the reduction of seed sources as a result of past logging over extensive areas. A further example of human intervention in natural processes can be found in the role, which introduced fauna species, including ungulates now play in altering or inhibiting successional pathways in many forest types in North America, Australia, New Zealand and Europe.

Although there is a hierarchy of scales of processes and factors that affect forest landscape patterns, in which factors at the upper level impose constraints on the next, there are also interactions among these factors from each level. Humans appear to be altering landscape-level processes at intermediate scales by forestry practices and, to a larger extent, by modern silvicultural practices such as controlling fires, logging forests and planting forests of types not expected from natural succession They are also causing changes at very large scales by altering climate. The result is altered landscape patterns at all scales, which may have eventual effects on forest plant species and their survival. This remains an important topic for research.

Forest landscape patterns are quantified by remote sensing from aerial photograph or satellite image information that has been entered into a geographic information system (GIS), and analysed using any one of several software tools, such as FRAGSTATS (McGarigal and Marks, 1995) and Patch Analyst (Elkie *et al.*, 1999), to quantify landscape variables. Common variables used include amounts of forest types by patch size and age-class, edge to interior ratio, fractal dimension, distance between patches, amount of non-forest, total edge, shape index, degree of interspersion and road density. The value of these statistics is that they enable comparisons between disturbed and undisturbed landscapes (or any two landscapes) under null models of intrinsic spatial patterns. Results of such comparisons can lead to comparisons and hypotheses with respect to biological diversity at large scales.

D. FACTORS LIMITING THE GLOBAL KNOWLEDGE OF FOREST BIOLOGICAL DIVERSITY

Several factors limit the global knowledge of forest biological diversity including insufficient taxonomy, a poor understanding of traditional knowledge, insufficient autecological and synecological knowledge of species, communities and ecosystems and the lack of infrastructure and capacity in many countries for inventory and monitoring programmes. Monitoring of biological diversity is a hierarchical procedure requiring a range of technologies and information. The scale can be depicted in terms of knowledge from forest types down to genetic resources with the associated required technologies and research to accomplish and develop the knowledge base. Where an individual country or agency is along that scale depends on its history of conservation policies, the dedication of the management agency to sustainable use, the availability of funding and the technical capacity and research available.

This chapter suggests that there is the need for a broad science research programme and data collection programme to understand the mechanisms affecting forest biological diversity and the relationship between biodiversity and forest goods and services. The research and management agenda, located in Annex lII, provides a path towards the sustainable management of forest resources.

There is a large number of sources of information pertaining to forest biological diversity, a partial list of such sources is located in Annex IV.

II. OVERVIEW OF FUNCTIONING OF FOREST ECOSYSTEMS AND RELATED GOODS AND SERVICES

A. INTRODUCTION

Forest ecosystems provide a wide array of goods and services at a range of scales from local to global. As well as sustaining commodities such as timber, traditional goods for local populations and services with economic return such as eco-tourism, forests perform a key role in providing vital services, which usually have no clear market value, notably global climate regulation and watershed protection. (Box 4).

The first part of this chapter aims to identify the key functional mechanisms for each of the three main forest biomes (boreal, temperate and tropical) and how they relate to biodiversity. It then assesses how human activities affect ecosystem functions and biodiversity and the ability of forests to deliver goods and services.

The second part of the chapter assesses the values to people of forest goods and services that are linked to forest biological diversity and the impacts of human activities on those values. There are many different perspectives on forest values, so it is important to distinguish these and analyse the implications of changes in forest biological diversity for the range of stakeholder interests.

Box 4: Examples of goods and services from forest ecosystems

Goods:

Timber and wood products, fuelwood and charcoal, non-wood forest products (such as bushmeat, medicinal and food plants) and biochemicals (for development of new medicines, etc.).

Services:

Watershed and water quality protection, conservation of biological diversity and genetic resources, purification of air, climate amelioration including carbon sequestration, soil conservation, religious, cultural, spiritual and psychological values and recreational and scenic values including ecotourism.

B. ....F UNCTIONING OF FOREST ECOSYSTEMS AND IMPACTS OF HUMAN ACTIVITIES ON RELATED GOODS AND SERVICES

i. Elements of forest ecosystem functioning, keystones species, functional groups notion of resilience

Understanding the role of biodiversity in the functioning of ecosystems is a relatively new field of research and many key aspects of ecological processes still need to be understood. Ecosystem functioning denotes the sum total of processes operating at the ecosystem level, such as the cycling of matter, energy and nutrients, as well as those processes operating at lower ecological levels which impact on patterns or processes at the ecosystem level (UNEP, 1995). Sound ecosystem functioning depends on the maintenance of a whole range of interactions between biotic and abiotic components. In turn, goods and services provided by forest biological diversity depend upon the maintenance in time of such sound and healthy ecosystem functioning.

In the context of massive biological extinctions driven by human impacts on land use, species invasions, atmospheric and climate change (WCMC, 1999), there is an urgent interest in understanding how loss of biodiversity affects ecological processes and, in turn, the functioning of the ecosystems. Many studies have focussed on the key ecological processes such as primary productivity, nutrient cycling and microbial activities. It appears that in most biomes, primary productivity seems to be weakly related to the number of plant species, although little to no information exists for forests (UNEP, 1995). However, diversity may play a role in the maintenance of productivity in the context of human induced and natural changes (UNEP, 1995; Chapin *et al.*, 1998). Some results show that the form and cause of the relationship between species and diversity and productivity may be highly dynamic, changing over time and space (Cardinale *et al.*, 2000).

Ecosystem functioning can be affected by the presence of keystone species and functional groups. Keystone species are defined as any which have an important effect on the community or ecosystem by virtue of unique traits or attributes (UNEP, 1995). The removal of keystone species can therefore result in dramatic changes in the functional properties of the ecological system. A functional group is a set of species that affect or control an ecosystem-level process in similar way (UNEP, 1995). A functional group can, for instance, be plants that have the same functions in the ecosystem. The concept of keystone species is an attractive one as it suggests potential indic ators and perhaps targets for management effort. However it is still a relatively untested idea.

Resilience quantifies the speed to return to equilibrium after a perturbation and is closely connected to ecosystem functions (Mooney *et al.*, 1996). More biolo gically diverse forests are generally thought to be more resilient and less liable to be affected by major outbreaks of pests and diseases.

ii. Boreal Forest Biomes

Overview of the functioning of the ecosystem

The boreal forest region is a relatively recent formation (<10,000 years). The forests are characterised by low richness of tree species. However, this low richness suggests potentially important functional roles for individual tree species (UNEP, 1995; Pastor and Mladenof, 1992). Strong feedbacks between species life traits, resources and disturbance regimes may cause cyclic fluctuations in animal populations (Hansson, 1979; Haukioja *et al.*, 1983).

Effect of species and genetic diversity on forest ecosystem processes

Functional mechanisms of biodiversity and functional groups

The primary functional grouping is conifers vs. deciduous hardwoods, and within this group, there are also species differences that cause a wide range of functional diversity. The low species richness, the important contrast among species regarding their traits and the low redundancy within each functional group is a major characteristic of boreal ecosystem (Pastor and Mladenoff, 1992; Cohen and Pastor, 1995). For example, the genera *Betula* and *Populus* represent early successional broadleaf species, *Abies* and *Picea* are shade tolerant conifers, whereas species of *Pinus* run the range from extremely shade intolerant to moderately shade tolerant. Thus, the removal of any single species from the ecosystem may have an important effect on the forest ecosystem functioning.

Plant tissue chemistry also represents a key functional attribute in boreal forest ecosystems, which integrates biodiversity with ecosystem properties. Tissue chemistry (particularly concentrations of nitrogen, resin, secondary compounds and lignin) controls decomposition and nutrient availability, palatability to herbivores and flammability and is also correlated with other functional plant traits such as life forms, growth rates and longevity (Pastor *et al.*, 1996).

Most of the boreal birds are migratory. Some groups include several specialised feeders that can exert considerable influence on their insect prey, such as warbler species on spruce budworm outbreaks in North America (Pastor *et al.*, 1996). The diversity of boreal mammalian fauna is intimately related to the history and diversity of trees. It is generally accepted that species richness decreases with increasing latitude, but recent studies have also shown a longitudinal gradient in the species richness of herbivores with the region near the Bering Sea being particularly species poor (Danell *et al.*, 1994) The fact that this region supports the woody species most chemically defended against browsing suggests that such gradients of plant chemical defence in boreal forests may be partly responsible for gradients of mammalian species richness (Pastor *et al.*, 1996).

Outbreaks of phytophagous insects can be significant regulators of forest primary production by releasing understory trees of species or age classes not susceptible to the outbreaks, and by returning nutrients to the forest floor (Mattson and Addy, 1975). Phytophagous insects such as spruce budworm (*Choristoneura fumiferana*), sawfly (*Neodiprion sertifer*), loopers (many genera), and bark beetles (e.g., *Dendroctonus ponderousae, Ips typographus*) are common major sources of tree mortality and morbidity in boreal regions. They typically show large oscillations in population levels, with spreading outbreaks occurring approximately ever few decades (Pastor *et al.*, 1996). Because such outbreaks occur periodically, defoliators and bark beetles play a major role in maintaining variability and diversity at the landscape scale by preventing the attainment of stable equilibrium communities (Pastor *et al.*, 1996).

Ants also represent an important component in forest ecosystems. The genera *Formica* and *Campanotus* form colonies that can have strong direct and indirect influences on nutrient cycling (Rosengren *et al.*, 1979).

The extreme fluctuations of animal population are among the most striking features of the boreal forest. Such fluctuations represent a temporally dynamic aspect of biodiversity. The dominant cycle length for a wide variety of mammals and birds in North America appears to be about ten years, while in Fennoscandia its length is usually four years (Keith, 1963; Finerty, 1980; Erlien and Tester, 1984). Cycles of herbivores may result in differential survival of their preferred food species, such as balsam fir, aspen and birch, as well as their predators, such as warblers that prey on budworm, or the Canada lynx that preys on small mammals (Keith, 1963; Stenseth, 1977; Hansson, 1979; Haukioja *et al.*, 1983; Bryant and Chapin, 1986; McInnes *et al.*, 1992).

Disturbances and species richness, effect on biodiversity

Natural disturbances, such as fires and insects outbreaks, are major processes maintaining the coexistence of species and diversity in the boreal zone. Fire is the dominant form of disturbance in boreal forest everywhere and individual fires can be very large (>several 100,000 ha), However most are smaller and spatial variation in intensity, site and regeneration responses also produce patch responses at smaller scale (10-1000 ha) within major disturbances (Baker, 1989; Payette *et al.*, 1985, 1989; Johnston, 1992; Mladenoff and Pastor, 1993). Fires generally occur in mature stands and in most areas they represent the predominant mechanism of forest regeneration.

By interacting with hydrological processes, the beaver (*Castor canadensis*) is an important agent of disturbance in boreal ecosystems and is considered to be a keystone species. Beaver ponds and wetlands can cover at least 13% of the land area in boreal landscapes (Johnston and Naiman, 1990). In North America, beavers are one of the major factors creating patches in the forested landscape (Hammerson, 1994).

Impact of human activities and their consequences on the delivering of goods and services

It is possible to consider two main types of human activities in the boreal biome: those that interact globally and those that interact locally with the ecosystem. Examples of activities with a global interaction include global warming and air pollution, including acid rain.

Global warming caused by anthropogenic emissions of greenhouse gases (such as carbon dioxide and methane) into the atmosphere, represents probably the greatest threat to the boreal region worldwide (Pastor *et al.*, 1996). It is generally agreed that the greatest warming will take place in high-latitudes regions, and particularly in the boreal zone. Although the exact extent and speed of the warming is still to be determined, the boreal climate change will dramatically affect the ecosystem in various ways. Some reports suggest a general northward shift of forest ecosystems (Emanuel *et al.*, 1985; Thompson *et al.*, 1998). However, the warming might be more rapid than the migration rates of major tree species, which may result in massive regional extinctions (Davis and Zabinski, 1992). In general, boreal tree species in the south will be replaced by northern hardwoods or prairies and the probability of fire will be increased over much of the biome. Also the probability of outbreaks of insect pests may increase as trees become drought-stressed and more susceptible

Anthropogenic depositions (SO_x, NO_x etc.) have an impact on water quality, tree health and soil properties in some regions. Although the trends and the mechanisms are not entirely clear (Manion and Lachance, 1992), the changes in soil properties (pH, CEC) could have a negative impact on the functioning of forest ecosystems and thus on biodiversity.

Human activities that have an impact on boreal ecosystems at local and regional scales include resource management activities such as logging and forest management, fire management and hunting.

Intensive hunting of fur-bearing mammals since the 16th and 17th centuries has led to a permanent reduction in the populations of brown bears (*Ursus arctos*) and martens (*Martes americana, Martes martes, Martes zibellina*) in their southern range, with consequences on their prey and on the overall ecosystem (Pastor *et al.*, 1996). The extermination of beavers from some areas of their previous range removed an important factor in landscape dynamics (Naiman *et al.*, 1986). However, most boreal mammals species are not currently endangered except for furbearing species that have been over-exploited, predators considered to be pests, or animals requiring large undisturbed areas such as wolves (*Canis* spp.), brown bears (*Ursus* spp.) and woodland caribou (*Rangifer tarandus*). Indeed, because of the generally great resilience of boreal species (Holling, 1992), most reduced animal populations rebound rapidly once direct exploitation stops (Broschart *et al.*, 1989).

Logging of natural primary forest may have major effects on the biodiversity of boreal ecosystems, unless old forests are permitted to redevelop (UNEP, 1995). Commercial logging currently dominates human-induced changes in boreal forests. As a result of logging, old growth stands have disappeared from large areas of boreal forest. Intensive forestry practices that consist of large-scale removal of natural mature stands and regeneration/replanting with an even-aged monoculture, often with soil preparation and artificial fertilisers, have had many negative ecological effects (Pastor *et al.*, 1996). More sustainable forest management activities are being introduced in Scandinavia and Canada with the aim of mimicking natural fire and wind disturbance patterns. These practices use patch felling, or small clearcuts with the retention of some old and dead trees as a ‘biological legacy’ [reference needed?]. The effects on biodiversity and ecosystem function of these ‘sustainable forest management’ systems are not yet known, but they are being practised in an adaptive manner.

Changes in fire regimes through active suppression of wildfires, partly as a result of forest management, have altered landscape-scale patch structure and age class distributions, as well as composition and stand structure (Heinselman, 1973; Baker, 1989). These changes interact with ecosystem processes to alter future successional pathways and forest productivity further (Pastor *et al.*, 1996). In turn, these changes in habitat distribution have an impact on biodiversity, particularly on bird populations (Pastor *et al.*, 1996).

Containing 23% of the overall terrestrial carbon stock and 49% of the total carbon stock of the three main forest biomes, the boreal biome represents the primary terrestrial carbon pool, far ahead of tropical and temperate biomes (UNEP, 2000). More than 84% of the carbon contained in boreal forest is stored in soils (UNEP, 2000). In addition to being the largest terrestrial carbon pool, boreal and temperate forest biomes represent a net sink and compensate for the net emissions from tropical ecosystems (UNEP, 2000). In this respect, the boreal biome plays a key role in climate regulation and any major disturbance to the ecosystem through human activities that cause a major release of boreal carbon may have significant consequences for global climate. Over the past century, Canadian boreal forests show a sharp rise in growth rates consistent with rising temperatures. Change in age class structure suggests they have gone from being a sink (about 0.2Gt C per year) to being neutral or a slight source of carbon to the atmosphere (Kurz *et al.*, 1995; Walker *et al.*, 1999).

iii. Temperate Forest Biomes

Overview of the functioning of the ecosystem

On a large geographic scale, the natural biodiversity of deciduous forests is the result of the distribution of species along different moisture and temperature gradients. At site level, only one or a few species gain dominance (Schulze *et al.*, 1996). These trends are also reflected in relation to elevation. Temperate forests are dominated by deciduous species and to a lesser extent by evergreen broad-leaf and conifer species. More than 1200 tree species are represented in this biome (UNEP, 1995).

Soil texture and fertility (determined by the base saturation) is a major factor in determining temperate forest biodiversity at site level. Soil acidity and nitrogen are important in determining the pattern of biodiversity, with more species growing on neutral or alkaline soils than on acid soils (Schulze *et al.*, 1996). Under natural conditions, the highest diversity is reached at high base saturation and high nutrient levels (Schulze *et al.*, 1996). However, decreases in woodland plant species diversity have been observed as a result of anthropogenic nitrogen deposition, for example in parts of Europe. Although high nitrogen deposition could potentially offset harvesting losses, it is also likely to exacerbate the acidification of soils (Schulze, 1989).

Effect of species and genetic diversity on forest ecosystem processes

Effect on productivity and on nutrient cycling

There is no evidence for an effect of biodiversity on net primary productivity in semi-natural forest stands. Productivity responds to available light and nutrients regardless of the number of species present in the stand (Schulze *et al.*, 1996). Some authors have shown that productivity is instead related to soil fertility and stand age. The factor most strongly correlated with productivity and nitrogen cycling is soil texture (Aber *et al.*, 1991) and the correlation is stronger in stands that have never been harvested. This suggests that disturbance by human activities can affect biogeochemical cycles by changing soil properties (Schulze *et al.*, 1996).

Effect of biodiversity on forest resilience

In general, greater species diversity increases the resilience of ecosystems against external disturbances (Tilman and Downing, 1994). In a stand containing diverse species, the loss of one species can be compensated for by the remaining ones. For example, the extinction of *Castanea dentata* in eastern North America (caused by a fungus) left a gap, which was filled by *Quercus* spp., suggesting redundancy within a diverse system. The same situation applies in the case of *Ulmus,* which was largely eradicated by Dutch elm disease in Europe, Canada and the USA, and was then replaced by *Acer*, *Fraxinus* and other species (Schulze *et al.*, 1996).

Forest resilience appears to diminish with growing anthropogenic effects such as air pollution and nitrogen deposition. Conifer stands in some parts of Europe, were seriously affected by anthropogenic pollution caused by sulphur and nitrogen deposition (Schulze, 1989). *Abies alba* showed signs of decline, but there was no loss of stand productivity where it grew in close association with *Fagus sylvatica,* which compensated for the loss. *Abies alba* disappeared over vast areas, apparently without any major effect on ecosystem functioning. In the case of *Picea* decline, *Fagus* and *Quercus* also showed decline symptoms and increased susceptibility to parasite damage (Schulze *et al.,* 1996), which had more obvious effects on forest structure.

Impact of human activities and their consequences on the delivering of good and services

Only 3% of old growth temperate forest remain, and in many regions temperate forests have a history of management and exploitation which has occurred over a large proportion of their area. Europe has a particularly long history of human use of forests and there are very few, if any, truly natural forests remaining. Many large predators have been lost from large areas for centuries, and domestic and wild grazing animals often have a strong influence on woodland composition and structure (Mayle, 1999). Woodland biodiversity has become adjusted to the traditional management and land-use patterns, with a higher proportion of species associated with open and edge habitats and younger growth stages, leading to more generalist species than in the past. However, historical management systems, notably coppice, wood pasture have declined over recent decades and a high forest structure is becoming more predominant. Ceasing management altogether, especially in fragmented woodland areas, is therefore likely to lead to loss of biodiversity and also other goods and services, such as recreation values, which depend on maintaining a range of age classes and open and wooded areas. Therefore, although there is a need for minimum intervention in some protected forest areas, (and there will be a place for new and restored natural forest in which the aim is to allow development with little intervention) the need in much of the temperate forest in Europe is to adjust forest management to modern conditions, rather than question whether to manage at all.

Effect of management on ecosystem functions

Different types of management systems can have different types of impacts on the forest ecosystem functioning. For example, patch felling may have a stronger local impact compared to selective logging, especially on soil fertility, as organic matter is more rapidly decomposed. Also, patch felling produces gaps in the forest, creating an edge effect and modifying the micro-climatic conditions: the air and soil without the shade and protection of trees tend to be dryer and warmer [reference needed?]. However, selective logging and patch felling at scales and intensities suited to the regeneration ecology of the main tree species are generally not disruptive to the ecosystem and should not significantly affect the future potential to supply most goods and services. Larger scale clear-felling, for example over areas of several hectares, causes substantial fluctuations in microclimate and soil and species composition. Logging roads can have a strong impact on the overall forest ecosystem (an impact which can be even greater than the tree harvesting itself), especially when layout and use do not take into account the restrictions of topography, soils conditions and hydrology.

The conversion of broad-leaf forest into coniferous plantations to increase timber production rates was prominent during the twentieth century in deciduous forests in some temperate countries. In these countries, large scale conversion began more than 100 years ago but its intensity is now reduced with new approaches advocating greater use of broad-leaf and native species. For example, *Fagus sylvatica* forest was massively converted to *Picea abies* plantations in Germany and deciduous forests in Japan are still being converted into *Cryptomeria japonica* in the south and *Abies* and *Picea* in the north (Schulze *et al.*, 1996). Nearly 40% of the ancient semi-natural broad-leaf woodland in the UK was converted to conifer plantation between around 1930 and 1985; after 1985 policy shifted towards conserving and restoring native woodlands (Kirby *et al.*, 1989).

The impacts of such changes on ecosystem functioning are important. The change from deciduous forest to conifer species often results in a decrease in the cycling of nitrogen between plant canopies and the forest soil, which can be 75% or more (Schulze *et al.,* 1996). The immediate consequence is increased loss of nitrate to ground water, which can affect water quality, notably where management involves large scale clear felling. There may be a long-term risk of soil acidification (Tsutsumi, 1977). Restoration of acidified soils often implies liming, which can have further impact on ground water quality by increasing nitrogen leaching.

Other impacts of coniferous conversion on diversity and related goods and services include reduced light penetration associated with evergreen foliage, which maintains drier soil conditions and this, together with the reduced litter decomposition rates caused by the chemistry of the needles, acts to inhibit the herb layer (Schulze and Gerstberger, 1993) at least during the younger stages of the forest rotation.

Fragmentation

Forest fragmentation can lead to an increase in species-richness through edge effects, but often this is achieved by adding generalist woodland/agricultural edge species at the expense of the loss of forest interior specialists (Andren, 1994).

Agriculture, urbanization and road building have substantially altered temperate forest integrity and ecosystem functioning. Fragmentation increases ecosystem vulnerability to other impacts such as invasive species, drift of fertilisers and herbicides from agricultural land, and climate change, which may cause localised extinctions of vulnerable species in future. Roads, including logging roads, introduce corridors, which promote the introduction of invasive plants, insects and pathogens (Schulze *et al.*, 1996), in addition to fragmenting the natural habitat for wildlife and modifying the drainage pattern. Studies have shown a correlation between the density of forest roads and the decrease of some animal populations (Brocke *et al*., 1989). Furthermore, poorly designed or operated logging roads can have a strong negative impact on the quality of the water as a result of erosion (Aber *et al.*, 2000).

Effect on carbon sink capacity

Temperate forests, along with the forests of the boreal biome, represent an important terrestrial net carbon sink (net uptake of carbon) as forest regrowth absorbs carbon dioxide from the atmosphere, where absorption rates exceed respiration rates. This compensates, to some extent, for the net emissions resulting from land use changes in the tropics (UNEP, 2000). This situation is explained by land use practices, natural regrowth, the indirect effects of human activities (e.g., atmospheric CO_2, fertilisation and nutrient deposition) and changing climate (UNEP, 2000). Expected growth in plantation areas will absorb more carbon, but the likely continued rate of deforestation will mean the world's forests remain a net source of CO_2 emissions and a contributor to global climate change (WRI *et al*., 2000). A plantation forest, managed for maximum volume yield, will normally contain substantially less carbon than the same area of unmanaged primary forest (Cannell, 1999). Multi-purpose planted forests, where some areas are allowed to mature and remain unharvested for biodiversity and other benefits should increase carbon sequestration compared to single purpose timber plantations and provide the best overall mix of goods and services.

Water distribution and quality

Watershed deforestation and subsequent inappropriate agriculture practices and soil erosion have had important effects on water quality and water quantity of streams and rivers. In some mountainous regions watershed protection is regarded as the main function of forests, for example in parts of Austria, Switzerland and Japan.

iv. Tropical Forest Biomes

Overview of the functioning of the ecosystem

One of the most distinctive characteristics of tropical forest is their biological richness, particularly in the number of species per unit of area (UNEP, 1995). In addition, for still unknown reasons, natural forests are relatively resistant to invasion by alien species. Invaders tend to be restricted to disturbed areas (Rejmanek, 1998; Whitmore, 1991). The complexity and diversity of tropical forests, especially humid forests, can be matched only by the underwater diversity associated with some coral reefs (Longman and Jenik, 1974). However, understanding the functioning of tropical forests is not a simple task because of the large spatial scale of the forests, the long time scale of their evolution and the high diversity of organisms in the forest fauna and flora. Climate and soils are the main factors controlling the distribution and the composition of tropical forests. A third factor is the interactions of their biotic components and with human activities (Longman and Jenik, 1974).

Effect of species and genetic diversity on forest ecosystem processes

Productive capacity, decomposition and nutrient cycling

In tropical moist forests, the number of species within a functional group greatly exceeds the number of key ecological processes, and even highly fragmented and disturbed forests have more species than the minimum to yield full primary productivity (UNEP, 1995). Thus, in tropical forests, biomass production under relatively constant conditions is insensitive to species richness (Orians *et al.*, 1996). However, some authors suggest that species richness may influence the rate at which biomass accumulates after a disturbance (Denslow, 1995). There is no inter-annual accumulation of litter in tropical soil, as the decomposition processes are too rapid (Barrow, 1991), thus soil degradation occurs very rapidly following the removal of the forest cover. Soil degradation is further enhanced in humid regions through runoff caused by heavy rains.

The case of mangroves

The intertidal forested wetlands, known as mangroves, support a vast amount of biodiversity in the tropical estuarine ecosystems. Mangroves also play an important role in maintaining water quality and shoreline stability by controlling sediment distribution in estuarine waters. Although there are relatively few species of mangrove trees (see Chapter I, paragraph 86), mangrove ecosystems are unique because they include structural niches and refugia for numerous animal species (UNEP, 1995). For example, crabs, mainly represented by two families - Grapsidae (63 mangrove species) and Ocypodidea (over 80 species) - are very abundant in mangroves systems and are considered to be keystone species. They positively influence tree productivity and reproduction, presumably by aerating the soil through their burrowing activities (Smith *et al.*, 1991).

Role of mobile species in forest biodiversity

Mobile species, such as pollinators and seed dispersal agents, have an important role in tropical forests. Many plants depend on a small suite of frugivores, such as bats ands birds, for dispersing their seeds, and loss of those species can have major influences on the long term population dynamics of many tree species (Orians *et al.*, 1996). In turn, loss of tree species may affect pollinators and dispersal agent diversity.

Functional properties over long temporal scales

The functioning of a tropical ecosystem depends on the formation and maintenance of the structure in the forest, which is the result of photosynthesis and biogeochemical cycling over a long time frame (many decades or even centuries) (Orians *et al.*, 1996). Certain life forms, such as the one represented by palms, lianas and epiphytic bromeliads, can be considered as “structural keytone species” because their removal would influence the rate of recovery of the forest after perturbations (Denslow, 1996).

Undisturbed tropical forests are resistant to invasion by exotic species. The mechanism of this resistance is unknown, however (Rejmanek, 1998).

Impact of human activities and their consequences on the delivering of good and services

Biodiversity and loss of habitats

Human impact on tropical forest biodiversity is mainly through land use change by converting forest areas to pastures, croplands, and plantations (UNEP, 1995) or by agricultural use and subsequent degradation and erosion. Arid and savannah forests are also threatened by desertification, which is exacerbated by overgrazing. The extent to which human activities are leading to the extinction of species in tropical forest is poorly documented, but current habitat and species loss in moist tropical forests are believed to be elevated and more important in tropical regions than anywhere in the world (Whitmore and Sayer, 1992). Such losses are expected to influence functional properties of tropical ecosystem (Orians *et al*., 1996).

Forest fragmentation, deforestation, reduction of habitat size and edge effects all have important effects on biotic linkages and on ecosystem functioning. Many tropical plants are animal-pollinated (Bawa and Hadley, 1990) and depend on animals for the dispersal of their seeds (Estrada and Flemming, 1986). As a result of forest fragmentation, the loss of pollinators or other functional guilds, such as seed-dispersing birds and bats, may affect the plant reproductive biology, forest structure and dynamics; and it may also affect the long term population of many tree species (UNEP, 1995; Howe and Smallwood, 1982; Terborgh, 1986), all having an impact on potential goods and services to humans.

Forest management in the tropics often consists of selective harvesting of commercial trees. Such trees can often occur at low densities. As in the case of selective harvesting in temperate forests, building of logging roads in tropical forests has an important negative impact on overall ecosystem functioning of the forests as the roads disrupt streams, lead to soil erosion, provide access to humans and open gaps in the canopy. Furthermore, logging roads in primary tropical forests often lead to poaching, uncontrolled settlements and illegal deforestation. However, in terms of providing goods and services, studies have demonstrated that secondary forests can be managed to provide many of the products that small-farmer households traditionally obtained from primary forests, while providing some of the environmental benefits of primary forests (Chapin, 1998 [or Chapin *et al.*, as in the list of references?).

Mangrove ecosystems are also under threat due to human activities such as land use change. Vast areas of mangroves are converted to other uses. For example, Thailand and Indonesia have lost 50% of their original mangrove ecosystems, Philippines 80% and Malaysia 32% (UNEP, 1995). Indirect causes of biodiversity loss include human alteration of upland watershed causing changes in fresh waters pathways or pollutions (UNEP, 1995). Climate change is also expected to have a direct dramatic effect on mangroves through the sea level rise.

Carbon pool

Tropical forests still represent the second largest pool of terrestrial carbon after the boreal forests (UNEP, 2000). However, from 1980 to 1998, net emissions of carbon dioxide into the atmosphere have occurred, mainly from land use change in the tropics (UNEP, 2000). It is interesting to observe that this net tropical emission is partly balanced by a net carbon uptake by the vegetation in middle and high latitudes, as a result of land use practices and natural regrowth (UNEP, 2000). While carbon fluxes might be weighed together in this way, the loss of biodiversity from tropical forest cannot, of course, be offset by any gains in other biomes.

Water distribution and quality

Watershed deforestation, inappropriate agriculture practices, and soil erosion have dramatic effects on water quality of streams and rivers. In addition, silt particles, which are carried to coastal zone, may cause death to coral reefs (UNEP, 1995).

v. Restoration of functioning of forest biological diversity in degraded forests or deforested lands

Forest restoration and re-afforestation of former forestlands is growing in importance. Two examples illustrate the range of these projects worldwide.

Watershed rehabilitation

One of the main problems linked to deforestation and, therefore, changes in forest ecosystem functioning is the alteration in the quality and quantity of water that drains through the ecosystem. The situation in the Himalayan foothills is particularly interesting. Because of the massive deforestation for firewood and agriculture needs, forests have been cleared over large areas in the Indian State Forest of Khol-Hai-Dun (Sommer, 1999). As a result, the soil has been heavily eroded by tropical rains and it has become unsuitable for agricultural practices. Also, siltation has affected streams, rivers and lakes. Rehabilitation programmes have been implemented through restoration of forest and grass cover and modification of agricultural practices.

Restoration and expansion of woodlands for biodiversity and other goods and services

In western Europe, for example in the United Kingdom, restoration of degraded ancient woodlands is becoming widespread, with the replacing of exotic species (which were planted during the 20th Century to boost wood production) with native species in an attempt to restore their characteristic biodiversity.

Both in Europe and in North America (and in the southern hemisphere) the forest area in many countries has increased considerably during the last 150 years through planting and by natural colonisation of abandoned farmland and industrial/mining areas. The current ecological value of these new woodlands from this period depends on their history (afforestation of agricultural land, heathland and eroded areas), management and actual forest functions. Where development to more natural woodlands is accepted or promoted (and especially in countries with low forest cover) they can contribute to a valuable increase in biodiversity.

Afforestation of agriculture lands with new ‘natural’ woodlands is becoming more popular, for example in the UK and the Netherlands. There is a programme of ‘new native woodlands’ where the aim is to develop woodland ecosystems that approach the original natural ones in their composition and functions and characteristic biodiversity, within the constraints imposed by losses of key species in former centuries and the irreversible naturalisation of some alien species. It is still unclear how long it will take for these woodlands to approach the biodiversity of surviving remnant natural forest. In the United Kingdom, it may take 200 years or more in some lowland arable landscapes, but 100 years or less in some upland pastoral landscapes, to allow slow colonists to cross hostile habitats. However many common species arrive much more quickly, perhaps within 50 years (Peterken and Game, 1984; Rodwell and Patterson, 1994). Much depends on the degree of linkage to existing old woodlands.

Planted forests established on agricultural lands, even those composed of exotic species, can eventually acquire significant biodiversity value, in combination with timber and other objectives (Cannell, 1999). Their potential contribution as part of landscape and regional scale strategies for conserving FBD should be given more attention in the future, as it is likely to be necessary for planted forests to continue to expand to relieve the pressure on natural forests for timber and fuelwood. They may help to buffer core natural forests against fragmentation effects and also help sustain larger populations of many of the more adaptable species .

Plantations and areas of natural regrowth have increased recently in some tropical areas, as well as in temperate regions. In Cuba, for example, forest area has increased from 14% to 21.1%, between 1959 and 2000, due to forest plantations and also to recovering natural forests [Modesto Fernandez Diaz-Silveira, pers. comm.].

vi. Assessing Status and Trends of forest ecosystem functioning

There is very little, if any, literature on assessing status and trends of forest ecosystem functioning. Identification, refinement and monitoring of various indicators is desirable to detect status and trends of forest ecosystem functioning in relation to goods and services. Indicators at the three biodiversity levels (genes, species and ecosystems) can be used to assess forest biodiversity status (see Chapters I and IV). However, indicators are still being refined and more work is needed to develop reliable and cost effective ways to monitor biodiversity.

For natural forest regions being opened up for the first time a simple and practical means of assessing forest ecosystem functioning could be found through the monitoring of macro-indicators, such as density of forest roads and socio-economic surveys. This information could be easily collected and analysed. The density of the forest road network could give a good indication of the fragmentation of the forest ecosystem, its "penetrability" and how intensively the forest is managed and which natural resources are used. Socio-economic surveys, on a local or regional basis, can give accurate information on the goods and services provided by the forest resources and their associated effect on the economy and on the benefits to the society. The correlation between these two indicators could provide a general idea on status and trends of the quality of the ecosystem functioning and its capacity to provide goods and services to the society.

vii. Conclusions concerning ecosystem functioning and human impacts

Human impacts on forest ecosystem functioning are numerous. Each of the three main forest biomes has it's own characteristics and, thus, the consequences of human activities within them will differ (see Table 6). It appears that in low-diversity systems, such as in the boreal forests, the full set of species within the functional groups may be important. In high diversity systems, such as the tropical forests, the deletion of species may sometimes be compensated for by other species. However this compensation does not seem to occur when all of a functional group is deleted (UNEP, 1995).

The boreal forest biome is characterised by relative low species richness, and by extreme contrasts in the functional attributes of important species for the ecosystem processes (Pastor *et al.*, 1996). Therefore, the loss of a few species can have significant impact on the ecosystem. Human activities such as logging and those that cause climate change may have a dramatic impact on the overall ecosystem functioning, which can, in turn, affect the delivery of good and services. Boreal forests represent 49% of the total vegetation and soil carbon contained in the three biomes and, for this reason, they play a key role in global climate regulation.

The effects of biodiversity on temperate forest ecosystem functions are indirect and interact through the chemical composition of foliage (C/N ratio), which affects decomposition (Schulze *et al.*, 1996). Biodiversity in temperate forest is determined to a major extent by the base saturation of the soil with the highest diversity being reached at high base saturation and high nitrogen availability. However, human-induced changes, such as land conversion, fragmentation and air pollution, are major factors in decreasing the diversity in deciduous forests and affecting their associated ecosystem functions (Schulze *et al.*, 1996). Climate change is likely to interact with these to cause further changes. Temperate forest in some regions has been exploited and managed for many centuries as part of cultural landscapes, and some form of continued management will be required for most areas to maintain a desirable range of ecosystem goods and services including characteristic biodiversity. Reducing fragmentation is vital in this biome, to buffer forests against impacts and insufficient or insensitive management

As explained in paragraph 0, overall the temperate biome is an important terrestrial net carbon sink. It is however unclear how long this situation will last as a saturation effect is expected (UNEP, 2000).

The main characteristic of tropical forest ecosystems is their biological richness and, unlike in boreal forest ecosystems, the number of species greatly exceeds the number of key ecological processes. This situation gives these ecosystems an apparent stability and probable resistance to invasive species. Tropical forests are also characterised by the very slow pace of their evolution, which is one of the reasons why it is difficulty to study and understand the ecological processes occurring within them. Thus, the consequences of the removal of a species by human activities may not be immediately observed. In addition to containing an important proportion of the world biodiversity, tropical forests also hold more than 37 % of the world's terrestrial carbon. However, because of deforestation and land use change, tropical forests represent a net source of carbon dioxide in the atmosphere.

Critical levels of biodiversity loss/change which can affect forest ecosystem functioning and, in turn, the goods and services provided by forests, are still hard to discern, and this needs to be a focus for future work. Keystone species or structures and functional groups need to be identified and validated in order to develop reliable indicators. While it is likely that some degree of biodiversity loss in some situations will have little or no long term effect on other goods and services, as long as the forest remains relatively intact, the linkages between biodiversity and ecosystem functions and the critical thresholds of impacts on biodiversity loss must be understood. This reinforces the value of following the precautionary principle when there is a reasonable doubt about the impacts of human activities on a forest ecosystem.

Most studies have focussed on loss of natural forest to unsustainable exploitation. There is a need to focus more on the potential at regional and landscape scales for synergy from combining primary and secondary natural forest, agro-forest and plantations on former open (non-forest) lands managed sustainably for a range of different packages of goods and services, including biological diversity. The application of the ecosystem approach advised by CBD should be the key way forward.

Table 6. Summary table on key ecosystem functioning, the principal human impacts and possible consequences on goods and services

Forest biomes	**Examples of human activities that can affect the ecosystem functioning**	**Key aspect of the ecosystem affected**	**Goods and services affected**	**Medium to long term possible main consequences**
Boreal	Clear cutting, harvesting and replanting	Rejuvenation of forest stand, age class variation, diversity.	Global climate regulation. Perturbation of the boreal forest ecosystem on large scale can affect the carbon uptake and storage capacity. Modification of landscape patterns.	Release of soil carbon.
	Fire control	Modification of stand structure and composition, regeneration, changes in wildlife habitat.	Landscape, wildlife populations.	Changes in species composition.
	Hunting	Impact on keystone species such as beaver, moose and on landscape pattern.	Landscape, wildlife populations.	Changes in species composition.
	Atmospheric pollution	Alteration of nutrient cycling, impact on water cycling, impacts on the overall ecosystem functioning, direct damages to the conifers.	Water quality, changes in forest composition, wildlife modification through habitat changes.	Changes in forest ecosystem composition.
	Fragmentation by silvicultural practices (patches)	Stand structure, overall ecosystem functions, wildlife habitat.	Wildlife populations.	Changes in wildlife species, migrations.
Temperate	Atmospheric pollution	Alterations in nutrient cycling, alteration of forest resilience.	Forest diversity, changes in forest composition and impacts on the food chain.	Health of the overall ecosystem. Increased susceptibility to outbreaks of insects and diseases.
	Sustainable forest	Patch, group or selective logging and	Timber, amenity, recreation can all	Low levels of old growth and biodiversity

Forest biomes	**Examples of human activities that can affect the ecosystem functioning**	**Key aspect of the ecosystem affected**	**Goods and services affected**	**Medium to long term possible main consequences**
	forest management systems	logging and regeneration: structural and species composition, few old/dead trees; roads; water and soil.	recreation can all benefit, and impacts on water soil and carbon may be small if well planned. Some aspects of biodiversity may benefit, others may suffer.	growth and biodiversity of dead wood. Biodiversity of young and pre-mature stages and open areas benefit. Most other goods and services sustained.
	Conversion of broadleaf forest to plantations of conifer species	Stand diversity, soil fertility, water.	Landscape, fauna and flora habitat, recreation and tourism, water quality.	Changes in species composition.
	Plantations on agricultural land or forest regeneration on agricultural land	Development of new woodland ecosystem	All main goods and services can develop to some degree.	May take 100-200 years or more for full development; may not be same as existing forest [Check details]
	Forest fragmentation by urban and infrastructure development.	Overall ecosystem functions affected.	Landscape, fauna, flora. Recreation and tourism.	Loss of habitats for fauna.
Tropical	Deforestation	Pollinator keystone species populations affected (e.g. bats and birds) and impacts on regeneration of pioneer species and ecosystem succession. Soil structure-texture	NTFP* , fuelwood, genetic resources, timber. Global climate regulation. Tropical forests are currently a net source of carbon dioxide. Eco-tourism. Water quality and forest regeneration affected	Loss of potential valuable genetic resources for agricultural and pharmaceutical use. Aggravating factor in the climate, global warming by release of carbon through burning and affecting the carbon uptake capacity of the forest. Soil erosion, soil sterilisation.

Forest biomes	Examples of human activities that can affect the ecosystem functioning	Key aspect of the ecosystem affected	Goods and services affected	Medium to long term possible main consequences
		Water regulation functions	affected Watershed protection	Flooding, soil erosion, impact on dam reservoir capacity, coastal pollution and impact on marine life. Loss of agricultural soils.
	Hunting	Impact on keystone species and their role on forest ecosystem regeneration.	NTFP*	Change in forest structure as seed dispersers such as primates and ungulates are removed
	Selective harvesting	Stand structure, Forest resilience	NTFP	Changes in fauna and flora, long-term consequences uncertain.
	Plantations on recently cleared tropical forest	Dramatic changes in the overall original ecosystem functions	NTFP, biodiversity water regulation.	Dramatic changes in local forest conditions, fauna and flora.

*Non-timber forest products (includes wild animals, fish, birds, honey, nuts, gums, fruits, flowers, spices, plants for local medicine use, fodder for animal, berries, mushrooms, etc.).

C. THE VALUE OF FOREST ECOSYSTEMS

This section analyses forest values from an economic perspective to explore how far the values of forests can be adequately integrated into markets, economic appraisals and decision-making in order to achieve sustainable use of forests and forest biological diversity. Justifications for a focus on economic value include the widely observed fact that forest conservation has ultimately to compete with alternative uses of forest land such as agriculture, agri-business, energy investments, roads and logging. Whereas these uses have reasonably clear and identifiable market values, many forest values are non-marketed. In a market-oriented world, therefore, forest conservation can easily lose out to the market values of alternative land use. However it must be recognized that there are inherent difficulties in applying economic techniques to values that are non-consumptive and in a manner that fully reflects the interests and perspectives of all types of stakeholder.

i. Types of forest values

Forests worldwide generate a wide range of goods and services that benefit humankind. From an economic perspective these values can be conveniently classified as:

(a) Direct use values: values arising from consumptive and non-consumptive uses of the forest, e.g. timber, fuel, bush meat, food and medicinal plants, extraction of genetic material and tourism.

(b) Indirect use values: values arising from various forest services such as protection of watersheds and the storage of carbon.

(c) Option values: values reflecting a willingness to pay to conserve the option of making use of the forest even though no current use is made of it

(d) Non-use values (also known as existence or passive-use values): these values reflect a willingness to pay for the forest in a conserved or sustainable use state, but the willingness to pay is unrelated to current or planned use of the forest.

There are other notions of value, for example, moral or ethical value, spiritual and religious value and cultural value. Moral and ethical values tend to relate to 'intrinsic' qualities of the forest and are generally not subject to quantification. The same is true of spiritual and religious values whereby forests embody characteristics venerated by individuals and communities. There are, however, links between these notions of value and economic value. In particular, non-use values are known to reflect many different motivations, motivations that include the individual's concern for intrinsic values. But notions of value based on intrinsic qualities are different to economic values in that the latter are always 'relational', i.e. they derive from human concerns and preferences and are therefore, values conferred by human beings.

ii. Stakeholders and Forest Values

Stakeholder analysis analyses the individuals, groups and institutions with an interest ('stake') in forests, assesses the nature of that interest, the impacts that such stakeholders have on forest integrity and ways in which those interests can be served in a sustainable manner. Table 7 sets out the classification of forest values and the interests that various stakeholders have in those values.

Table 7. Forest values and stakeholder interests

Forest value	Main stakeholders and their interests	Impacts on forest integrity
Direct use values		
Timber	Logging companies (profit) Government (royalties)	Often unsustainable Usually low tax-take
Fuelwood	Local* communities (high value)	Usually sustainable
NTFPs	Local communities (high value)	Usually sustainable
Genetic information - Agriculture - Pharmaceutical	Plant breeding companies (profit) Drugs companies (profit) Local communities (medicines)	Sustainable [1] Sustainable Sustainable
Recreation	Tourists (revenue leakage issue)[2] Nearby urban dwellers	Usually sustainable Sustainable?
Research/education	Local and international universities	Sustainable
Cultural religious	Local communities	Sustainable

Indirect use values *Watershed functions* Soil conservation Water supply Water quality Flood protection Fisheries *Global climate* Carbon storage Carbon fixing Biodiversity Amenity (local)	 Local and regional communities Local and regional communities Local and regional communities Local and regional communities Local and regional communities Global community[4] (climate protection) Local community (carbon trades) Global community (climate protection) Global community Local communities Nearby residents	 Usually unappropriated[3] Usually unappropriated Usually unappropriated Usually unappropriated Usually unappropriated Favours conservation Favours conservation Favours conservation Favours conservation Favours conservation Unappropriated benefit[5]
Option and existence values	Global community debt for nature swaps, donations, forest funds, GEF etc) Local and regional communities	Appropriable Not usually appropriated
Land conversion values Crops Pasture Logging Agro-forestry Agri-business Aquaculture (mangrove)	 Agriculturists Ranchers: Local communities Private businesses Logging companies Governments Local communities Private companies Private companies Local communities	 Inconsistent with forest conservation Inconsistent with forest conservation Generally unsustainable Potentially sustainable Inconsistent with forest conservation Usually inconsistent with mangrove conservation

Notes: * In all cases 'local' is meant to include indigenous communities wherever appropriate
1 Forests probably not the main source of agricultural genetic material.
2 Revenues accrue to international companies or companies away from forest area, little revenue accrues to local communities.
3 Sustainable indirect uses for which no market transaction usually exists, but where market creation is possible (see experience of Costa Rica).
4 World population, especially those in climate-vulnerable areas, plus all institutions (CBD, FCCC etc) seeking climate protection, plus NGOs.
5 i.e. residents not charged for benefit.

An important feature of Table 7 is that forest conversion values can accrue to local communities (e.g. shifting agriculture) but that such practices are increasingly unsustainable as less open access forest is available. The effect of the 'diminishing frontier' is that fallow plots are revisited before regeneration has fully occurred, so that second and third round crop production takes place on increasingly 'mined' soils. Indigenous peoples and local communities may benefit at least in the short term from other conversion activities, e.g. employment from logging operations. Often, however, the converted land use involves ownership by other agencies, e.g. national or regional government and larger corporations, with the effect of displacing local communities. For indigenous peoples this can also create dependence on the monetary economy and trigger far-reaching social and cultural disruption, without opportunities for earning money.

Table 7 also shows that local communities already benefit substantially from forest goods and services. In particular, fuelwood and other NTFPs can account for a substantial fraction of local community income. Communities could benefit further from the monetization of carbon storage and sequestration flows through private carbon trades and/or trades as envisaged in the flexibility mechanisms of the Kyoto Protocol. The same is true of market creation in watershed protection benefits, as shown in Costa Rica's Forest Law of 1996, and in the formalisation of intellectual property rights in genetic information under the Convention on Biological Diversity. Local communities might therefore be beneficiaries of processes designed to appropriate the benefits from forest non-market values. [Check comments]. The inverse of this proposition is also true - they are likely to be the major losers from processes that continue to convert forest land. However, there are many potential negative impacts with these flexibility mechanisms, such as displacement of indigenous peoples and local communities from their lands, forest destruction, denial of land and land use rights, commercialization and monetization without corresponding development opportunities.

Development of a CBD action plan for FBD should include a thorough stakeholder analysis at global level, in order to ensure that the interests and potential contributions of different key groups and organizations are appropriately and fully taken into account.

iii. Economic values

Direct use values

The value of forests is most commonly associated with the production of timber and fuelwood. These are major products for many countries, providing building materials, energy, pulp and paper, industrial raw materials and valuable foreign exchange. Estimates by FAO (2001b) show that global production of roundwood reached 3335 million m^3 in 1999, a little more than half of which is used for fuelwood and and the remainder for industrial roundwood.

Timber values

Two types of timber use need to be distinguished: commercial and non-commercial. Local uses may be commercial or can relate to subsistence, e.g. building poles. World industrial roundwood production expanded substantially between 1960 and 1990 from some 1 billion m^3 to 1.6 billion m^3 but has since fallen back to some 1.5 billion m^3 in the late 1990s (Barbier *et al.*, 1994; FAO, 2001b). Tropical wood production in 1999 represented a relatively small proportion of overall global production of the various commodities: about 15% of the world’s industial roundwood production, 14% of sawnwood, 15% of wood-based panels and 9% of paper and paperboard (FAO, 2001b). Industrial roundwood production in 1999 was dominated by developed countries, which together accounted for 79% of total global production. Industrial roundwood production varied from year to year during the 1990s, but the overall trend was relatively flat. This was a significant change from the rapid growth that occurred prior to 1990. Wood-based panel and paper/paperboard production show a steadily rising demand, which is partially offset by reductions in the demand for sawnwood.

Fibre production has risen nearly 50% since 1960 to 1.5 billion m^3 annually. In most industrial countries, net annual tree growth exceeds harvest rates; in many other regions, however, more trees are removed from production forests than are replaced by natural growth. Fibre scarcities are not expected in the foreseeable future. The potential for forest plantations to partially meet demand for wood and fibre for industrial use is increasing. Although accounting for only 5% of global forest cover, forest plantations were estimated in the year 2000 to supply about 35% of global roundwood, with anticipated increase to 44% by 2020. In some countries, forest plantation production already contributes the majority of industrial wood supply (Carle *et al.*, 2001).

In a comprehensive survey of sustainable forestry practice, Pearce *et al*. (2001) found that *sustainable* forest management is less *profitable* than *non-sustainable* forestry, although problems of definition abound. Profit here refers only to the returns to a logging regime and do not include the other values of the forest. Sustainable timber management can be profitable, but conventional (unsustainable) logging is more profitable. This result is largely due to the role that discount rates play in determining the profitability of forestry. The higher the discount rate the less market value is attached now to yields in the future. If logging can take place in natural forests with maximum harvest now, this will generate more near-term revenues than sustainable timber practice. Similarly, sustainable timber management involves higher costs, e.g. in avoiding damage to standing but non-commercial trees. The significance of the general result is that the non-timber benefits, including ecological and other services, from sustainable forests must exceed the general loss of profit relative to conventional logging for the market to favour sustainable forestry.

Fuelwood and charcoal

FAO (2001b) statistics suggest that in 1999 some 1.75 billion m^3 of wood was extracted from forests for fuelwood and conversion to charcoal. Of this total, roughly one-half comes from Asia, 26% from Africa, 10% from South America, 8% from North and Central America and 5% from Europe. The International Energy Agency (1998) estimates that 11% of the world's energy consumption comes from biomass, mainly fuelwood. IEA (1998) estimates that 19% of China's primary energy consumption comes from biomass, the figure for India being 42% and the figures for developing countries generally being about 35% (see also UNDP *et al.*, 2000). All sources agree that fuelwood is of major importance for poorer countries and for the poor within those countries. While fuelwood may be taken from major forests, much of it comes from woodlots and other less concentrated sources. Extraction rates may or may not be sustainable, depending on geographic region. Hardly any fuelwood and charcoal is traded internationally.

As with other non-timber products (see below), local values of fuelwood and charcoal can be highly significant in terms of the local economy. Shyamsundar and Kramer (1997) show that the value of fuelwood per household per annum for villages surrounding Mantadia National Park in Madagascar is \$39. This can be compared with an estimated mean annual income of \$279, i.e. collected fuelwood from the forest accounts for 14% of household income. Houghton and Mendelsohn (1996) find that the value of fuelwood constitutes from 39-67% of local household income from fodder, fuel and timber in the Middle Hills of Nepal.

Non-timber forest products

NTFP extraction may be sustainable or non-sustainable and few studies make observations as to which is the case. One example of sustainable use is in the Sinharaja Forest Reserve in Sri Lanka, where the most popularly collected NTFPs (*Calamus* species/rattans, *Caryota urens*/kithul palm used for jaggery production, wild cardamom and a medicinal herb, *Costcinium fenestratum*) all performed better in selectively logged-over forest than in undisturbed forest, where they were either absent or showed poor growth (Gunatilleke *et al.*, 1995).

Extractive uses include: taking mammals, fish, crustaceans and birds for local or international trade or for subsistence use, taking plant products such as latex, wild cocoa, honey, gums, nuts, fruits, flowers/seeds, berries, fungi and spices, also plant material for local medicines, rattan and fodder for animals. Detailed analysis of the available studies suggests that economic values for NTPF (net values, i.e. net of costs) cluster from a few dollars per hectare per annum up to around US$100/ha/yr. Lampietti and Dixon (1993) suggested a 'default' value of around US$70 per hectare, and Pearce (1998) has suggested US$50[10]. However, these values cannot be extrapolated to all forest. Typically, the higher values relate to readily accessible forests, values for non-accessible forests would be close to zero in net terms due to the costs of access and extraction.

The benefits of NTFPs accrue mainly to local communities. The size of the population base making use of the forests may be comparatively small and the implied value per hectare may therefore also be small due to the unit values being multiplied by a comparatively small number of households. It is important to discern, as far as possible, what the value of the NTFPs is as a percentage of household incomes. Available studies suggest NTFPs may account for 30-60% of local community household income and in some cases the amount exceeds 100% of other income. This perspective demonstrates the critical importance of NTFPs as a means of income support. Indeed, it underlines (a) the need to ensure that measurements of household income include the non-marketed products taken 'from the wild' and (b) the role that NTFPs play in poverty alleviation.

Biodiversity and genetic information

Tropical forests probably contain more than half the world's terrestrial species. Numbers vary according to whether mammals, birds, insects or plants are being considered. Islands have a critical role to play in biodiversity, often containing high species endemism. The economic value of this diversity arises from the fact that diversity embodies the value of *information* and *insurance*. Existing diversity is the result of evolutionary processes over several billion years and subject to many different environmental conditions. Hence, the diversity of living things also embodies characteristics that make them resilient to further 'natural' change (but not to many human interventions). In essence, the existing stock of diversity provides the entire range of goods and services, including information, provided by the diverse system [check comments].

The value of genetic variation can be expressed as ecological, economical or ethical value. In practice, however, it is difficult to determine whether a specific genetic form (variant) of a species will be of future value. Hence, within species, it is difficult to distinguish between genetic resources and genetic variation (Graudal *et al*, 1995). Potentially, the information embodied in biodiversity can be used in, for instance, plant breeding, into developing drugs and perhaps into industrial processes. The more distinctive the information is, the more potentially valuable it is, so that the existence of substitutes is a critical factor affecting the economic value of the information. This has affected efforts to value the information content in several ways. First, while forest degradation continues, it can be argued that the remaining stock is so large that willingness to pay to conserve part of the stock is currently small. That willingness to pay will rise as the stock depletes. Second, the willingness to pay will be small as long as there are substitutes and this is true of both agricultural germplasm and 'medicinal' germplasm. Also relevant is the fact that research and development effort is more easily diverted to genetic manipulation than to the identification of 'wild' genetic information.

Swanson (1997) reports the results of a survey of plant-breeding companies, finding that the sampled companies rely on germ-plasm from relatively unknown species for a small part of their research (i.e. on *in situ* and *ex situ* wild species and landraces). Swanson's analysis suggests that the stock of germplasm within the agricultural system tends to depreciate at a rate of about 8% of the material currently in the system. Thus the stock of germplasm within the agricultural system is being renewed at a time interval that is probably around 12 years (100/8). But the 8% comes from a stock of natural assets – biodiversity – that is itself eroding. Hence the loss of biodiversity worldwide imposes an increasing risk on the

[10] The values shown also reflect local market conditions and there is no reason why prices will be similar in the different locations, e.g. because incomes vary significantly.

agricultural sector. Biodiversity has economic value simply because it serves this maintenance function. Without it, there are risks that the system will not be able to renew itself.

There are several ways of estimating the economic value of this germplasm. First, it could be argued that the economic value of wild crop genetic material is what the crop breeding companies are willing to pay for it. At a minimum, this must be equal to that portion of their R&D budgets spent on germplasm from the more remote sources. Second, an effort could be made to estimate the crop output that would be lost if the genetic material was not available. This is an approach based on damages. Third, an attempt could be made to estimate the contribution of the genetic material to crop productivity – a benefits approach. This approach might proceed by asking what the cost would be of replacing or substituting the wild genetic material should it disappear – a ‘replacement cost’ approach.

The role of forests in providing agriculturally relevant genetic information should not be exaggerated. As far as plant based foods are concerned, existing widely used crops tend not to emanate from tropical forests but from warm temperate regions and tropical montane areas. The existing 'Vavilov' centres of crop genetic diversity are mainly in areas with low forest diversity. However, it does not follow that forests are irrelevant to future crop production. It seems probable that their value lies more at the regional than the global level (Reid and Miller, 1987). Overall, systematic estimates of the informational value of wild species to crop output are not available and remedying this is an important research task.

The informational value of forest diversity for pharmaceutical use is better studied. There is little dispute about the local values of traditional medicines, and these are substantial within the context of a local economy (see under NTPFs). There is more debate about the ‘global’ value of medicinal plant material. The economic studies are concerned with the values of marginal species [is meaning of this sentence clear?]. The total value of biodiversity is clearly unbounded; without biodiversity there would be no human life and hence no economic value. In the pharmaceutical context, the relevant economic value is the contribution that one more species makes to the development of new pharmaceutical products and, by inference, the value of one extra hectare of forested land is the value attached to the species in that area The ownership rights to this kind of information is a controversial issue and there are varied views on how to recognise the importance of indigenous peoples’ traditional knowledge and whether and how they should be compensated for the use of the information and genetic material derived from it.

The available studies that focus on the pharmaceutical value of forests focus on the 'hot spots' - areas of high endemic diversity that are heavily threatened. The evidence suggests that pharmaceutical genetic material could be worth several hundreds of dollars per hectare in most hotspot areas, and perhaps up to several thousands of dollars for selected areas. For the major part of the world's forests, however, values will be extremely small or close to zero. [reference needed?]

Diversity is a precondition for all the other values defined for the forest, from tourism to timber and non-timber products, and including the information flows. On this basis, the economic value of diversity as insurance is the premium that the world should be willing to pay to avoid the value of the forest goods and services being lost. The *actuarially fair premium* for this insurance, if a market for it existed, is the probability of the loss occurring multiplied by the value of all the losses that would occur (Pearce, 2001 or Pearce *et al.*, 2001). No attempt has been made anywhere to estimate, even approximately, what this premium is, but it is clearly very large since the probability of loss is known to be high[11] and the values are also potentially high. The complication, again, is that the premium will be small for the initial continuing losses of forest cover, rising only as the forest cover is lost.

Tourism and recreation values

Ecotourism is a growing activity and constitutes a potentially valuable non-extractive use of tropical forests. Caveats to this statement are (a) that it is the *net* gains to the forest dwellers and/or forest users

[11] And could be estimated from the current rates of loss of forest cover.

that matter; (b) tourism expenditures often result in profits for tour organisers who do not reside in or near the forest area, and may even be non-nationals; (c) the tourism itself must be 'sustainable', honouring the ecological carrying capacity of the area for tourists. In principle, tourism values are relevant for any area that is accessible by road or river. Some forest ecotourist sites attract enormous numbers of visitors and consequently have very high per hectare values. Values clearly vary with location and the nature of the attractions and none of the studies available estimates the extent to which expenditures remain in the region of the forest. For tropical forests, values range from a few dollars per hectare to several hundred dollars [reference needed?]. A substantial number of studies exist for the tourism and recreational value of temperate forests. Indicative values for European and North American forests suggest per person willingness to pay of around $1-3 per visit. The resulting aggregate values for forests could therefore be substantial. Elasser (1999) suggests that forest recreation in Germany is worth some $2.2 billion per annum for day-users alone and a further $0.2 billion for holiday-makers.

Indirect use values

Watershe d protection

Watershed protection functions include: soil conservation and hence control of siltation and sedimentation; water flow regulation, including flood and storm protection; water supply and water quality regulation, including nutrient outflow. The effects of forest cover removal can be dramatic if non-sustainable timber extraction occurs, but care needs to be taken not to exaggerate the effects of logging and shifting agriculture (Hamilton and King, 1983) or permanent conversion to agriculture. Available studies suggests that watershed protection values appear to be small when expressed per hectare, but it is important to bear in mind that watershed areas may be large, so that a small unit value is being aggregated across a large area. Secondly, such protective functions have a 'public good' characteristic since the benefits accruing to any one householder or farmer also accrue to all others in the protected area. Third, the few studies available tend to focus on single attributes of the protective function - nutrient loss or flood prevention etc. The aggregate of different protective functions is the relevant value. Fourth, the Hodgson and Dixon study (1988) for the Philippines suggests that fisheries protection values could be substantial in locations where there is a significant in-shore fisheries industry. Comprehensive estimates have still to be researched.

Carbon storage and sequestration

Brown and Pearce (1994) suggest benchmark figures for carbon content and loss rates for tropical forests. A closed primary forest has some 280 tonnes/ha of carbon and if converted to shifting agriculture would release about 200 tonnes of this, and a little more if converted to pasture or permanent agriculture. Open forest would begin with around 115 tC and would lose between a quarter and third of this on conversion. Using such estimates as benchmarks, the issue is what the economic value of such carbon stocks is. A significant literature exists on the economic value of global warming damage and the translation of these estimates into the economic value of a marginal tonne of carbon. A recent review of the literature by Clarkson (2001) suggests a consensus value of US$34/tC. Tol *et al.*, (2000) also review the studies and suggest that it is difficult to produce estimates of marginal damage above US$50/tC. Taking US$34-50/tC as the range produces very high estimates for the value of forests as carbon stores. In practical terms, however, a better guide to the value of carbon is the price at which it is likely to be traded in a 'carbon market'. Carbon markets have existed since 1989 and refer to the sums of money that corporations and governments have been willing to invest in order to sequester carbon or prevent its emission. More sophisticated markets will emerge as emissions trading schemes develop under the Kyoto Protocol. Zhang (2000) suggests that, if there are no limitations placed on worldwide carbon trading, carbon credits will exchange at just under US$10 per tC.[12] At this carbon 'price' tropical forest carbon storage would be worth anything from $500 per hectare to US$2000/hectare, confirming the view of a number of

[12] However it should be noted that current actual prices are often significantly less than this, and there are considerable political, legal, economic, social and technical obstacles to ensuring that the benefits of carbon sequestration are shared equitably.

commentators that carbon values could easily dominate the economic values of tropical forests. These sums are 'one off' and therefore need to be compared to the price that is paid for forestland for conversion to agriculture or logging. In most cases, carbon storage is more than competitive with conversion values. These values relate to forests that are (a) under threat of conversion and (b) capable of being the subject of deforestation avoidance agreements.

Carbon regimes in temperate countries have also been extensively studied and afforestation carbon values probably range from about US$100 to $300/ha.

Option and existence values

There are three contexts in which option and existence values might arise: (a) someone may express a willingness to pay to conserve the forest in order that they may make some use of it in the future, e.g. for recreation. This is known as an *option value*; (b) someone may express a willingness to pay to conserve a forest even though they make no use of it, nor intend to. Their motive may be that they wish their children or future generations to be able to use it. This is a form of option value for others' benefit, sometimes called a *bequest value*; (c) someone may express a willingness to pay to conserve a forest even though they make no use of it, nor intend to, nor intend it for others' use. They simply wish the forest to exist. Motivations may vary, from some feeling about the intrinsic value of the forest through to notions of stewardship, religious or spiritual value, the rights of other living things, etc. This is known as *existence value.*

There are few studies of the non-use values of forests. The available evidence suggests that (a) existence values can be substantial in contexts where the forests in question are themselves unique in some sense, or contain some form of highly prized biodiversity - the very high values for spotted owl (*Strix occidentalis*) habitats illustrate this; and (b) aggregated across households, and across forests generally, existence values are modest when expressed per hectare of forest. [check comments].

Valuing sustainable forestry

One approach to estimating the economic value of sustainable, as opposed to 'conventional' forest management, is to determine what consumers are willing to pay by way of a price premium for timber and wood products from certified forests. Certification schemes exist to guarantee the sustainability of certain forests, akin to ‘eco-labelling’ of products. Various certifying bodies are accredited by the Forest Stewardship Council (FSC) and 3.5 million m^3 of certified timber entered international trade in 1996, whilst 10 million ha of forests had been certified by 1998 (Crossley and Points, 1998). Certification costs are around US$0.2 to US$1.7 per ha for developing countries and 9-12 cents per acre for assessment and 1-3 cents per acre per annum for licensing and auditing in the USA (Crossley and Points, 1998). Accordingly, any willingness to pay by consumers above this level of cost represents the ‘net premium’ for sustainable forest management.

The evidence on the premium consumers are willing to pay (WTP) for certified timber is mixed. A survey of four studies in Barbier *et al.*, (1994) revealed the following:

(a) a survey of UK *manufacturers* in 1990 suggested 65% were WTP more for certified timber;

(b) ....a survey of UK *consumers* in 1991 suggested a 13-14% WTP premium;

(c) a survey of UK *consumers* in 1992 indicated that 58% would not buy timber if they knew it came from rainforests; and

(d) a 1992 survey of *timber importers* suggested that 70% thought their customers were not WTP for certified timber.

An additional survey in British Columbia suggested that 67% of respondents to a survey would pay 5% more for certified timber, and 13% said they would pay 10% more (Forsyth *et al.*, 1999). Crossley and Points (1998) suggest that certified products are securing premiums of 5-15% in some cases, but that the real benefits of certification for industry lie in securing greater market share and longer-term contracts. There is some evidence that companies gaining certification secure higher company value, i.e. the value of certification shows up in share prices on the stock exchanges.

Summary of economic values

Table 8. summarises the economic values outlined above. It is important to understand the limitations of the summarised estimates. Values will vary by location so that summary values can do no more than act as approximate indicators of the kinds of values that could be relevant. Nonetheless, the table suggests that the dominant values are carbon storage and timber. Second, these values are not additive since carbon is lost through logging. Third, conventional (unsustainable) logging is more profitable than sustainable timber management. Fourth, other values do not compete with carbon and timber unless the forests have some unique features or are subject to potentially heavy demand due to proximity to towns. Unique forests (either unique in themselves or as habitat for unique species) have high economic values, very much as one would expect. Near-town forests have high values because of recreational demand, familiarity of the forest to people and use of NTFPs and fuelwood. Uniqueness tends to be associated with high non-use value. Fifth, non-use values for 'general' forests are very modest.

Table 8.. Summary economic values (US$/ha/pa unless otherwise stated)

Forest good or service	Tropical forests	Temperate forests
Timber Conventional logging Sustainable Conventional logging Sustainable	 200-4400 (NPV)[1] 300-2660 (NPV)[1] 20- 440[2] 30- 266[2]	 -4000 to + 700 (NPV)[3]
Fuelwood	40	-
NTFPs	0- 100	small
Genetic information	0-3000	-
Recreation	2- 470 (general) 750 (forests near towns) 1000 (unique forests)	80
Watershed benefits	15- 850	-10 to +50
Climate benefits	360- 2200 (GPV)[4]	90 - 400 (afforestation)
Biodiversity (other than genetics)	?	?
Amenity	-	small
Non-use values Option values Existence values	 n.a. 2- 2 4400 (unique areas)	 70? 12 - 45
Sustainable forest management premium	5-15% of timber prices?	5-15% of timber prices?

Notes [1] See Annex 1.2 of background document - annuitised NPV at 10% for illustration.
[2] THIS IS MISSING
[3] – (Pearce 1994).
[4] - assumes compensation for carbon is a one off payment in the initial period and hence is treated as a present value. It is a gross value since no costs are deducted.
3 EXPLAIN NPV AND GVP IN TABLE 3

Emerging methods for valuing forest goods and services

Estimates of economic value of forest services tend to be based on a few economic valuation methodologies. The first of these is the 'production function' approach whereby some output or service is measured. The output or service is then valued at market prices (e.g. the price of timber, fuelwood, or medicinal plants, etc.). Some of these values can also be derived by stated preference techniques, notably contingent valuation. Stated preference techniques seek to elicit willingness to pay through the use of structured questionnaires. The advantage of these techniques is that they measure directly the total value that users of forest products are willing to pay for them. For tourism and recreation the most widely used technique is the travel cost method. This method looks at the expenditures made by people travelling to a forest site, using their expenditure as a means of estimating willingness to pay for the site experience. The valuation of genetic information has been based on what the purchasers (e.g. a drug company) of that information are, implicitly, willing to pay. In turn, this willingness to pay reflects the value of the genetic information as a potential input to the manufacture of the good in question (e.g. a drug). Hence the value of the genetic material is a 'derived' demand and reflects the production function approach again. The same is true of watershed values in that the forest, as an 'input' to watershed protection, defines the object of value. Avoided expenditures tend to be the source of the unit value, e.g. the willingness to pay of a hydroelectric company for upstream forest conservation reflects the losses that would otherwise accrue due to reservoir sedimentation if the forest was degraded. Climate benefits also tend to be based on the production function approach: climate regulation is an input to many services such as avoided sea level rise, crop damage etc. The individual forms of damage may be valued in many different ways, but market prices and avoided costs tend to dominate. Finally, non-use values can only be valued by stated preference techniques, i.e. through questionnaires about willingness to pay, because non-use values leave no 'behavioural trail' for the analysts to assess.

Table 9. gives valuation techniques applicable to some forest goods and services. One technique listed appears to have general application. This is 'choice modelling'. Choice modelling refers to a range of techniques in which respondents to a questionnaire are presented with options between which they have to choose. The options combine various features or attributes. The level of these attributes is varied across the options so that respondents are choosing between different 'bundles' of attributes. A price or cost is generally included as an attribute. Rather than stating their willingness to pay for the different attributes, respondents imply valuations through their rankings. The analyst elicits the valuations through econometric procedures. The relevance of choice modelling to the forest context will be evident. Potentially, each forest good or service can be treated as an attribute. The attributes will vary in level across different forest management systems (and across different forest conversions). In principle, then, choice modelling could lead to valuations of each of the attributes of the forest. To date, there have been only a few studies of forest values using choice modelling.

Table 9.. Valuation techniques for forest goods and services

Forest good or service	Valuation techniques					
	PF	MP	AC	CV	CM	TC	HP
Timber		√			√		
Fuelwood		√	√		√		
NTFPs		√		√	√		
Genetic information	√	√			√		
Recreation/Tourism					√	√	
Watershed	√	√	√		√		
Climate	√	√	√	√	√		
Biodiversity				√?	√		
Amenity				√	√		√
Non-use values				√	√		

PF = production function, MP = market price, CV = contingent valuation, CM = choice modeling
[WHAT’S AC, TC AND HP?]

Costs and benefits of forest conversion

Some attempt can be made to look at the likely costs and benefits of converting existing forests to alternative uses. Unfortunately, the data remain very limited and there is the added problem that costs and benefits will vary by forest location. It should also be noted that the comparisons all assume that non-market values are actually captured through some market creation mechanism. Bearing this in mind, the evidence suggests the following conclusions: 1) converting primary forest to any use other than agro-forestry or very high value timber extraction is likely to fail a cost-benefit test; 2) the conversion of secondary forest to the 'cycle' of logging, crops and ranching could make prima facie economic sense. As with the primary forest conversion, however, it needs to be borne in mind that the 'sequence' of land use does not always occur and many conversions to slash and burn agriculture would make no economic sense; 3) the conversion of secondary forest and open forest to agro-forestry appears to make economic sense assuming that most of the forest's services (including biodiversity) are retained; 4) carbon storage is of the utmost importance to the economic case for forest conservation; 4) the non-market values almost certainly fail to capture the economic value of biodiversity which, apart from the value of genetic information, is omitted from the analysis.

iv. The causes of forest loss

In this section, causes of forest loss are analysed from an economic point of view to examine how far they could be addressed by improving economic incentives for sustainable use and conservation of forests. However, it should be stressed that the limitations of market mechanisms need to be explored and recognised in relation to the needs of stakeholders. Market mechanisms must complement other mechanisms including legislation, regulation, certification, capacity building and addressing wider underlying causes of forest loss (see next section). A more comprehensive analysis of these issues is contained in Chapter III.

It is important to distinguish between the direct and underlying causes of forest loss. The main direct causes include unsustainable logging, slash and burn agriculture, the building of infrastructure such as dams and roads, pollution, fires, infestation and invasive species. However, statements about direct causes provide little insight into the issues that would have to be addressed by policy measures. For this it is necessary to ask why each of the direct factors comes about - e.g. why do loggers behave unsustainably, why do shifting cultivators behave as they do and so on. The basic concept of relevance is that of an *economic incentive* to engage in deforestation or forest damaging activities. These economic incentives are reinforced by, or embedded in, issues such as rapid population change, corruption and lack of information. In turn, however, it is important to ask what the incentives are for these contextual factors.

Missing markets

Probably the most important feature of forest goods and services is that many of them have no market. As such, there are no market forces to send the appropriate price signals to users of forest land that forests have economic value in conservation or sustainable use. The essential requirement is that conserved or sustainably used forest must secure returns to provide an economic justification for conservation. While it is not essential that these conservation values show up as cash flows, or flows in-kind, there is obviously more chance that conservation will occur if they do have associated real benefit flows. The overall conclusion is that, despite the early literature [some references needed?] suggesting non-timber benefits could greatly outweigh those from slash and burn and/or clear felling, sustainable commercial uses of forest land have considerable difficulty competing with alternative commercial uses such as conventional logging, agri-business and agriculture. There will be exceptions to this rule. Given the difficulties of competing, the importance of 'encashing' the other benefits of forests is to be emphasised, especially carbon storage and sequestration and, where relevant, tourism and the sale of genetic material.

Discount rates

One of the features underlying comparisons of the relative profitability of different forest land uses is the role of the discount rate. High discount rates favour conventional logging over sustainable timber management, slash-and-burn agriculture over agro-forestry and so on. The issue is therefore one of knowing how large discount rates are in such contexts. Existing research suggests that local communities often have high discount rates of well over 10% and up to 30 or 40%, reflecting their urgent need to address subsistence and security needs now rather than in the future (Poulos and Whittington 1999). While this conclusion should not be exaggerated - there are many examples of poor communities investing in conservation practices - the available evidence supports the traditional view that many have high discount rates and that these contribute to 'resource mining'.

Property rights

It is well established that the existence of complete, exclusive, enforced and transferable property rights is a prerequisite for the efficient management of natural resources. Rights must be complete and exclusive to avoid disputes over boundaries and access. They must be enforceable to prevent others from usurping them and they must be transferable (there must be a customary or full market in them) to ensure that land is allocated to its best use. The effects of incomplete or no property rights show up most clearly in the lack of incentive to invest in conservation and sustainable land uses. Regardless of the 'paper' designation of forest land rights, many forests are *de facto* open access resources, i.e. resources for which there is no owner**.** Other forests are common property and are managed by a defined group of households with rules and regulations about access, use and transferability. Provided common property resources are not subject to external forces that lead to the breakdown of the communal rules of self-management, common property is a reliable and reasonably efficient use of forestland. Factors causing common property breakdown include rapid population growth and interference in traditional communal management by central authorities. Traditional, customary and, sometimes, even legally recognized land rights of indigenous peoples can be hard to establish and are often ignored or violated.

Establishing property rights in the form of communal or private ownership regimes is a prerequisite to efficient land use, but may still not guarantee the desirable level of forest protection. This will be the case where the forest values take the form of 'public goods', i.e. services and goods the benefits of which accrue to a wide community of stakeholders and for which no mechanism exists to charge them for the benefits. Forest dwellers may then have no incentive to conserve forests for their benefits to downstream fisheries or water users, since they receive no benefit for these services. Institutional change designed to compensate forest users for these services can often be devised (see below), effectively establishing property rights in the unappropriated benefits of forest services.

There is a small but growing trend towards the redefinition of property rights in forests to take account of these factors. Carbon trading provides one clear example, whereby corporations or agents in one country invest in sequestration or conservation in another country in return for the paper credit certifying the amount of carbon so stored or sequestered. Probably the greatest progress in establishing property rights in forest services has been made in Costa Rica (Chomitz *et al*., 1998). Costa Rica's forestry law of 1996 recognises the value of forests as carbon stores, providers of hydrological services, protectors of biodiversity and providers of scenic beauty. Sources of finance, e.g. a fuel tax, were designed and the rules for paying forest owners for services were established. The Costa Rican government currently disburses money for reforestation, sustainable forest management and forest preservation. Landholders cede their rights to the relevant services to the National Forestry Fund (FONAFIFO) for five years in return for the payments. A summary of this project is in the Annex 5.

Perverse incentives

Governments worldwide provide incentive systems that affect natural resource use. While usually conceived with good intentions, they often have deleterious effects on natural resources. Notable examples include the $800 billion spent each year on subsidising certain economic activities, especially agriculture ($400 billion). Most subsidies are in the developed economies, where agricultural subsidies are responsible for some reduction in woodland area, the woodland being removed to capture the subsidies, which are often on a per-hectare basis [reference needed]. In some parts of the developing world subsidies exist for the clearance of forest land, and in some cases title to the land cannot be secured without a given percentage of the land being cleared [reference needed]. Other subsidies are more subtle, and may take the form of preferential logging concessions and low royalty rates relative to what could be charged without deterring logging companies. Low charges increase the 'rent' to be secured from the land. The result is a competition to capture the rent, a competition that uses up resources to no productive purpose. Ensuring a good share of rent capture can involve corrupt practices such as bribes to officials and politicians. In turn, this can result in more extensive logging outside 'official' concessions and more intensive logging inside concessions as those responsible for enforcement secure greater rewards from the bribes than they do from normal employment. Unsustainable logging is more immediately profitable and hence there is a financial incentive to override or ignore regulations designed to secure sustainable forest management. The extent of 'illegal' logging is not known with any accuracy but is clearly very large and may, in some countries, greatly exceed the officially declared rates of logging. Tackling illegal logging is immensely complex since it effectively involves tackling the corruption involved. Countervailing power in the form of NGOs and citizens' groups can help, as can a free media and international disapproval. Statistical studies suggest that political freedom may be linked to reduced deforestation, but the evidence is not firm (Kaimowitz and Angelsen, 1998). Overall, though, there are powerful incentives for illegal logging and deforestation generally.

Population change

Brown and Pearce (1994) reviewed the econometric studies that link deforestation rates to explanatory factors. They found that population growth is generally linked to deforestation, although the patterns of interaction are complex. However, though simple statements that 'population growth causes deforestation' are unquestionably false, many models show that population change is important (Kaimowitz and Angelsen, 1998). As current population levels rise from 6 billion people to a predicted 9 billion in 2050, with much of the increase in tropical countries, pressures on forest areas must be expected to grow. Lowland-upland migrations and officially induced transmigrations will add to the pressure.

Indebtedness

It is widely surmised that the more externally indebted a forested country is, the more likely it is to engage in policies that result in deforestation. This happens because there is pressure to export logs and processed wood and, to a far lesser extent, other forest products to secure foreign exchange to meet the interest payments on the debt. A number of econometric studies test this relationship and the balance of evidence suggests that there is some link between indebtedness and deforestation (Kaimowitz and Angelsen, 1998). Very few studies find any link between timber prices and deforestation, i.e. the expected relationship that logging will increase as world prices for the wood increase is not found.

Internal factors

A number of factors internal to the forested country may contribute to deforestation. In regions of primary forest, road building has the obvious effect of opening up forested areas. Logging roads will have this effect initially and subsequent hardcover roads may exacerbate the situation by encouraging agricultural colonists to enter the area. Satellite pictures identify 'leaf vein' patterns of land use following initial opening up for logging or for highways and studies using statistical models to test for the effects of roads suggest that this deforestation will occur (Kaimowitz and Angelsen, 1998). Anything that reduces transportation costs will also tend to encourage deforestation, which would previously have been limited by the costs of getting produce to the market. More generally, the closer forests are to towns the greater the risk that they will be subject to clearance [reference needed?]. It has been suggested that raising agricultural productivity will lower deforestation by reducing the incentive to 'extensify' agricultural land use. Again, only limited econometric evidence supports this hypothesis. Income levels should also be linked to deforestation, with higher incomes perhaps increasing deforestation initially and later reducing it. Poverty should be linked to deforestation on the grounds that open access resources add significantly to household income. Econometric studies tend to find that higher incomes increase deforestation, suggesting that the initial phase of the expected relationship is in place. Globally, forest cover is clearly linked to income since European and North American forest area is increasing.

Excessive consumption

As incomes rise, so the demand for natural resources increases. The relationship is a complex one, however. For some forest services, the income-demand relationship can be such that as incomes grow the demand for those services decrease. An example might be the switch from wood fuels to liquid fuels as incomes grow. At the global level, however, higher income countries do consume larger absolute amounts of raw materials. This has led to the view that deforestation is linked to 'excessive consumption' in rich countries. The issue is complex because the efficiency of raw materials use, i.e. the ratio of raw materials to income, tends to be lower in richer countries than in poor countries. Rich countries utilise natural resources more efficiently, but the scale of their incomes means that the absolute level of consumption is higher than in poor countries. Since the aim of development is to raise per capita income, reducing that income is not a realistic policy option, nor is it clear what policies would bring this about without damaging the factors giving rise to income growth - education, technology etc. But it is legitimate to ask that rich countries greatly increase their resource use efficiency. This will then translate into reduced demand for raw materials, including forest products imported from developing countries. Care has to be taken that this does not damage the export potential of forested countries, but clearly there is scope for making this transition. Additionally, richer countries can afford to pay premiums on forest products to discriminate between sustainably and unsustainably managed products.

III. MAJOR THREATS TO FOREST BIOLOGICAL DIVERSITY

A. INTRODUCTION

The previous chapters have shown that forest biodiversity is directly linked to the existence of forest and to the way forests are managed, and that deforestation and forest degradation themselves are the main proximate causes for loss of forest biodiversity. Therefore, in order to identify and propose measures to halt and reverse global forest biodiversity loss, the CBD will have to address both the direct and underlying causes of forest decline. Effective action will require an understanding of the underlying causes. It is often more immediately profitable to deforest areas or to log forests in an unsustainable way than to sustainably manage them, and this has been identified by the AHTEG as one of the primary causes for the high rate of deforestation and forest degradation and, therefore, for the current loss of forest biological diversity.

This chapter addresses the main threats to forest biological diversity and, to do so, the causes have been divided into direct (proximate) and underlying (ultimate) causes (see figure 1):

Direct causes

The direct (or proximate) causes of biodiversity loss in forests are human induced actions[13] that directly destroy the forests (such as conversion of forest land, continuous overexploitation or large scale logging) or reduce their quality (by, for instance, unsustainable forest management or pollution).

Underlying causes

The underlying (or ultimate) causes of forest destruction are the factors that motivate humans to degrade or destroy forests; complex causal chains are usually involved. The underlying causes originate in some of the most basic social, economic, political, cultural and historical features of society. They can be local, national, regional or global, transmitting their effects through economic or political actions such as trade or incentive measures (WWF, 1998).

The driving forces behind direct human impact on forest degradation and deforestation and, consequently, on biodiversity loss are both numerous and interdependent. The approaches needed to try and remedy adverse human impacts are country specific and will, therefore, vary between countries (WRI *et al.*, 1992; Mc Neely *et al.*, 1995; Contreras-Hermosilla, 2000; IFF, 2000; WWF *et al.*, 2000).

Interactions between direct and underlying causes

The interactions between direct and underlying causes are very complex: the cause-effect relationships will vary a lot from country to country and/or over time and there can therefore be no overall hierarchy between the causes; they do not interact linearly, but rather in a circular fashion with many feedback loops. Even a single force, such as agricultural intensification, may operate in a very different way under one set of circumstances than it would in a different situation with other variables involved. Accordingly, remedial measures need to be tailored to the very specific situation to which they will be applied. There are no simple solutions to this complex phenomenon. (CIFOR, 2000).

[13] Although natural factors (e.g. glaciations) can contribute to the destruction or degradation of forest ecosystems, in most cases, natural occurrences, such as fires, storms, volcanic eruptions, or insect outbreaks, do not destroy forests; they can even be part of their regeneration patterns. However, it is becoming more and more difficult to distinguish between natural and human induced factors. For instance, factors such as climate change (human induced) could contribute to an increase in the number and intensity of such so called ‘natural’ factors.

The distinction between direct and underlying causes of forest degradation is often not clear as it appears. In reality, there are long, complex causation chains that eventually lead to deforestation. Causes may be hierarchical. For example, a hypothetical chain of causes and effects may operate in this way: shifting cultivators deforest because they need to provide a means of survival for their families. This is because they are poor and have few alternatives to deforestation. They are poor because present power structures discriminate against a large number of people who therefore have little or no access to alternative means of survival. Present power structures originated in historical arrangements such as colonisation. Thus, in this theoretical example, there is a causation chain that starts with colonisation and runs through unequal control over key resources, to poverty and the need to survive and, finally, to forest decline.

In contrast to the example above, few cause-effect chains are linear or unidirectional. Instead, there are many branches that constitute secondary cause-effect loops leading to forest decline[14]. Feedback loops complicate analyses of the causes of forest decline. For example, a logging company may construct harvesting roads that facilitate the occupation of forestlands by small farmers. After some time, these farmers may be successful in lobbying politicians not only to improve these roads but also to build new roads, thus making it easier for new migrants to obtain access to forested areas located further away.

Causal factors are likely to vary over time, sometimes drastically. At certain stages of development, rapid income growth could promote forest decline by, for example, increasing demand for forest products and by enhancing human capacity to alter forests. When economies reach a certain threshold, the process is reversed. At this point, increases in the level of income per capita begin to be associated with factors such as technological improvements, better functioning of government institutions, urbanisation and less relative dependence on agricultural and forest production. That leads also a change in the composition of demand for goods and services with greater demand for environmental services of forests and for uses, such as recreation, that do not necessarily lead to the loss of forest cover (Contreras-Hermosilla, 2000).

[14] There are also some important feedback effects working in the opposite direction.

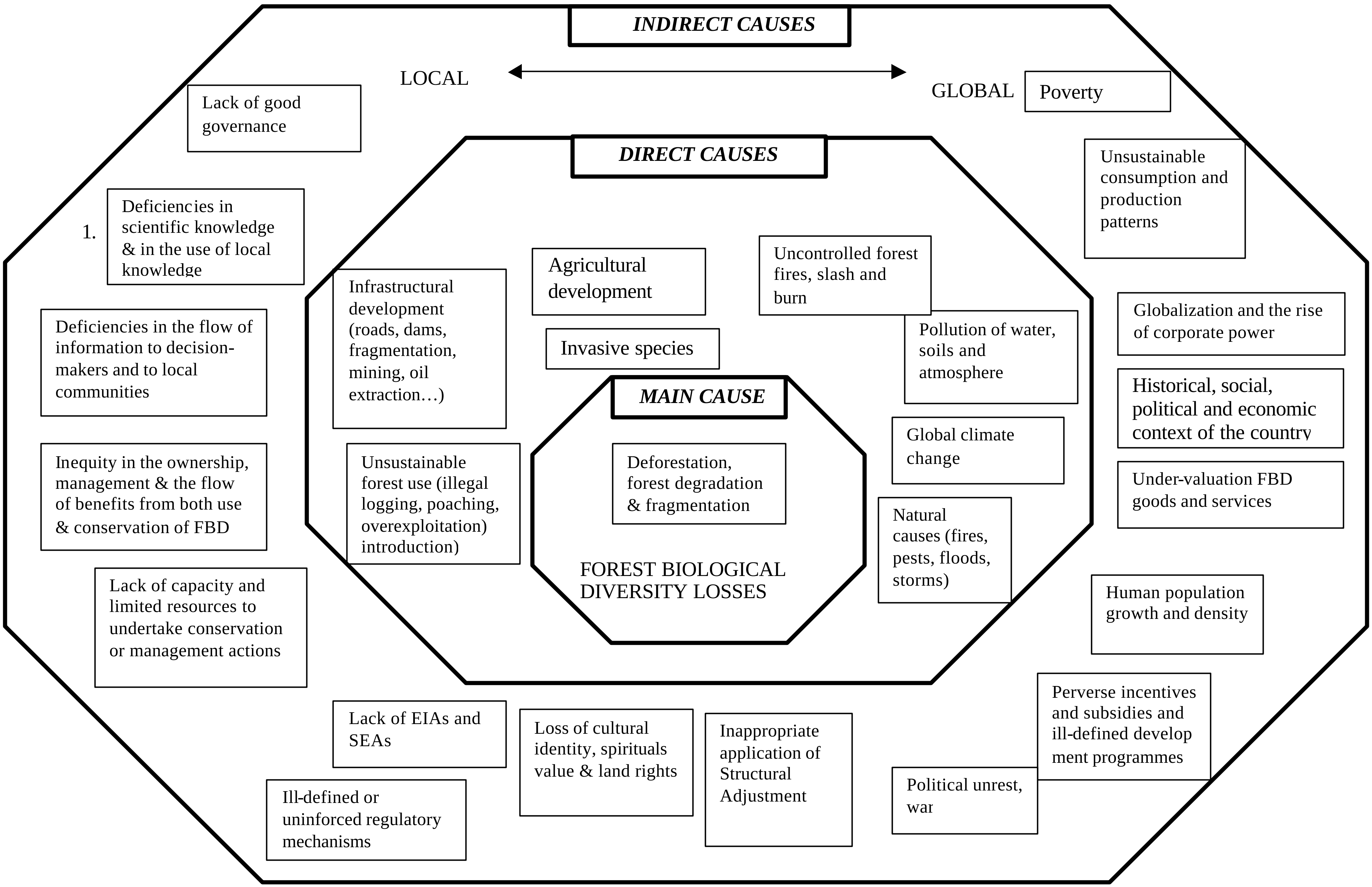

Fig. 1: Main causes of forest biological diversity loss (CIFOR, 2000)

B. AN HISTORICAL AND PRESENT PERSPECTIVE OF FOREST BIOLOGICAL DIVERSITY

Current concerns over the rates of loss of biological diversity as a consequence of human population growth and resource use have been well-documented (Pimm *et al.*, 1995, UNEP 1995, WRI 2000). However, loss of species, including forest dependent species, is an evolutionary fact, as evidenced through the fossil record, which suggests that 95 per cent of the species that have ever lived are now extinct (Raup, 1991). Several major periods of mass extinction are clearly shown in the fossil record, with the last mass extinction being about 65 millions years ago, at the end of the Cretaceous period. Aside from catastrophic mass extinction, a continuous background extinction rate has also been variously estimated, depending on taxon and species concerned. Interestingly, the background extinction rate appears to have declined for most taxa (Valentine *et al.*, 1991) except terrestrial flora, where the rate has increased (Niklas *et al.*, 1985). Numerous theories have been advanced to explain both background and mass extinctions and debate continues. However, species accumulation seems to occur in geologic time following each extinction event (Jablonbski, 1991).

While the causes of past extinctions remain under debate, the recent phenomenal rate of extinctions associated with humans has been substantiated for a few cases, some noted in Pimm *et al.* (1995). Humans have caused extinctions for several thousands of years (Martin, 1973). A particular example is the Polynesian occupation of Pacific islands, which resulted in the extinction of more than 2000 bird species [reference needed]. In the north-eastern United States and southern Canada, an area with trees was logged continuously since European colonisation. During that period, four of 26 endemic bird species (16%) have gone extinct and it has been calculated that humans now cause about 1000 - 10000 extinctions per million species-years (Pimm et al., 1995). Extinction is not always a rapid process. Many species may be driven to extinction over several hundred years by now immutable processes. However, in the 15th and 16th centuries, the global spread of European cultures and their livestock, crops, weeds and diseases increased the loss of island flora and fauna and added to the threats to continental species. In later centuries, the growth of trans-oceanic human travel and commerce led to the spread of a tremendous variety of species to new regions of the world and human colonisation of many uninhabited islands. Native species may be out-competed by the exotic species introduced, deliberately or accidentally, by humans. Current estimated rates of extinction for most higher life-forms in tropical rain forests are 1-10% of these species in the next 25 years (or 1000 to 10,000 times background rates) (UNEP, 1995).

Causes of extinction are many: Allee effects in small populations, environmental change, demographic and genetic stochasticity, excessive killing by humans, competition from exotic species, habitat loss and fragmentation and other habitat changes (Box 5). Most extinctions faced by contemporary animal species are the result of habitat loss and competition (or predation) by exotic species (WCMC, 1992). Indeed, habitat loss is the major threat to 75% of endangered birds (Pimm *et al.*, 1995). For trees, the number of species at risk is largely a result of logging and clearing for agriculture, which together threaten about 2300 species (Oldfield *et al.*, 1998).

Extinctions may be caused by isolation of populations, followed by inbreeding and loss of fitness to react to environmental changes, competition and natural selection (see Chapter 1). Lessons from population genetics indicate that in forest areas, large areas of habitat must exist to maintain populations of large-bodied species, there must be connections between habitat patches to enable gene flow, populations should always be larger than that which is minimally viable, there should be multiple separate populations to avoid loss by local catastrophic events, and great care should be taken with island ecosystems. Of course, in biological systems, cascading effects can result from individual extinctions, when a species with particular functional importance to an ecosystem (keystone species) is removed (see earlier chapters).

Some countries are attempting to redevelop historical forest types, based on retrospective studies of biodiversity and forest structures. The idea is to establish a 'benchmark' of forest types, structures and age classes in the landscape and then to try and redevelop historical ecosystems through silvicultural techniques. For instance, in eastern Canada, which has had a history of clearcutting and high-grade logging over 300 years, several important conifer species, such as white pine (*Pinus strobus*), red spruce (*Picea rubens*) and eastern hemlock (*Tsuga canadensis*) have been severely depleted. Redevelopment of the historical forest ecosystem, as part of the biodiversity objectives set for the forest, is occurring within the Canadian Model Forest Network Programme (see Box 9 in Chapter IV).

C. DIRECT CAUSES OF LOSS OF FOREST BIODIVERSITY

The direct cause of forest biodiversity loss is forest decline, in particular the destruction and/or degradation of natural and semi-natural forest ecosystems (WRI *et al.*, 1992; WCMC, 1992; Barbier *et al.*, 1994a; ESD, 1995; Stedman-Edwards, 1998). In general, all forest types are affected by deforestation and degradation. However, at a global level the forest associations which have already disappeared or are at most risk in the future are those occupying habitats most suited to agricultural development, notably lowland forests and woodland communities on flatter terrain with fertile soils. In addition, various riparian forest ecosystems have greatly diminished in extent and quality due to agricultural development, reduced water flows and/or inundation for dam development (irrigation and hydro-power) and severe flooding events due to catchment changes and/or climate change. Forests associated with certain mineral ore bodies have also suffered or remain at high risk, e.g. specialised floras on nickel- and heavy metal-rich ultramafic soils in New Caledonia and the Solomon Islands in the south-west Pacific.

As mentioned earlier, human actions are the most important direct cause of forest loss. Their destructive actions include: adverse agriculture practices (conversion to agricultural land and industrial agriculture, overgrazing and unmitigated shifting cultivation); unsustainable forest management (such as poor logging practices, over-exploitation of wood for fuel and charcoal manufacture, over collection of non-timber forest resources); introduction of exotic and/or invasive plant and animal species; infrastructure development (road building, hydro-electrical development, improperly planned recreational activities); mining and oil exploitation. In addition, human cause forest fires, pollution and climate change all of which are likely to lead to forest loss.

At the species level, extinction is commonly associated with extreme rarity, and insufficient reproductive capacity: the six main direct causes of species extinction being habitat loss and degradation, fragmentation, overexploitation, secondary extinction, introduction of exotic species and climate change (Diamond, 1984; Terborgh and van Schaik, 1997). Loss of genetic diversity in a species can be associated with loss of genetically distinctive populations, especially in those species with high levels of between-population variation. The main causes of local extinction or the loss of a particular population are the same as those for loss of a species.

i. Agriculture

Conversion of forests to agricultural land

The major causes of deforestation are the expansion of subsistence agriculture and large economic development programmes involving agriculture. The conversion of forests into agricultural land has been the major historical cause for deforestation in Europe and still is a major driving force in the tropical and sub-tropical areas. The agents vary from small farmers practising shifting cultivation or clearing forests for subsistence needs to large agricultural concerns that clear vast tracts of forest lands in order to establish cattle ranches or agro-industrial plantations such as soya beans in Latin America and oil palm in Indonesia/Malaysia.

Dismantling of agro-forestry systems

An emerging and rather insidious threat to biological diversity and tree genetic resources is posed through the dismantling of agro-forestry systems, i.e. the removal or failure to plant trees in agricultural and horticultural systems. This is usually associated with intensified, often monocultural, agricultural and livestock husbandry practices that eliminate trees from rural and urban agricultural areas. In Tonga, especially on Tongatapu, successive phases of unsustainable cash cropping have led to the elimination of trees in agro-ecosystems. In parts of Africa many useful tree species now exist only as scattered individuals or highly fragmented non-viable populations in agro-ecosystems, and are likely to disappear within the next few decades [reference needed]. Trees in agro-ecosystems may disappear either directly through cutting and clearing, or through the establishment of a hostile environment for regeneration and recruitment of remnant tree species.

Overgrazing

Overgrazing is increasingly a major threat to biodiversity in both tropical and temperate forests. The main impacts are damage to the topsoil, destruction of understorey vegetation and/or its replacement with a narrower range of unpalatable species and selective browsing of regenerating tree species, which may eventually result in the elimination of particular species.

ii. Unsustainable forest management

Poor forest utilisation practices remain widespread in all forest types, despite well-documented studies and widespread demonstrations of the environmental and economic benefits of sustainable management methods such as reduced-impact logging (RIL) practices (Putz *et al.*, 2000) or close-to-nature forestry.

Poor logging practices

Poor logging practices are those that lead to a longer-term reduction in the actual and potential economic, ecological and social functions of the forest. For example, large clear cuts using heavy machinery can lead to soil compaction and nutrient washout. Another form of poor logging practice is the destruction of specific habitats, such as rivers and streams. Poor logging practices adversely impact on biodiversity values of forests by degrading them in various ways, and sometimes make it more likely that the areas will be converted to an alternative land use at a later date. Several more sustainable logging practices exist.

Reduced impact logging

Reduced impact logging systems are currently being developed in several tropical countries in response to concerns over the ecological and economic sustainability of harvesting natural tropical forest stands. RIL systems use an array of best harvesting techniques that reduce damage to residual forests, create fewer roads and skid trails, reduce soil disturbance and erosion, protect water quality, mitigate fire risk, help maintain regeneration and protect biological diversity (Holmes *et al.*, 2000). Ecological damage is minimised by taking an inventory of trees before harvesting and cutting only mature trees.

Putz *et al.* (2000) have examined the various possible reasons why RIL practices are not being more widely applied. These authors conclude that the main reason for low adoption rates is that, under certain conditions, RIL adds to logging costs rather than delivering savings. These conditions include situations in which compliance with logging guidelines restricts access to steep slopes and/or prohibits ground-based timber yarding on wet ground. Putz *et al.* (2000) conclude that widespread adoption of RIL may require financial incentives, such as those which may be provided due to enhanced carbon sequestration in carefully logged forests c.f. conventional logging (e.g. Andrewartha and Applegate, 1999).

Over-intensive forest management

In temperate and boreal forests, several aspects of over-intensive forest management negatively impact on forest biodiversity. These include: the replacement of natural and semi-natural forests by monospecific and even-aged plantations, large scale clear-cutting in areas without natural large scale disturbances, the planting of inappropriate species, the removal of important forest structures such as dead and decaying wood, the destruction of key habitats and the disturbance of drainage patterns.

"Close-to-nature" forestry

However, in many countries of the temperate and boreal zones, forest management methods that try to maintain a high level of biodiversity in managed forests (e.g. "close-to-nature" forestry) have been increasingly applied in recent years. These management methods mimic natural disturbances regimes, use local and site adapted species, increase the diversity of species, ages and structures in the forest stands, maintain important micro-structures such as dead and decaying wood, protect important key habitats and limit the use of pesticides, fertilisers, drainage and damaging machinery. The Pro-Silva movement, an association of European foresters that advocates close-to-nature forestry, has developed specific principles and guidelines related to sustainable forest management, the conservation of ecosystems, the protection of soil and climate, the production of timber and other forest goods and the recreational, amenity and cultural aspects of forests (Box 6). According to the German National Forest Programme[15], close-to-nature forestry has been developed over the past few years in all states (Länder) largely relying on the use of natural processes and self-steering mechanisms. In many boreal forest countries (Canada, Finland, Norway, Sweden), private companies, state forest services and forest owners increasingly apply specific forest management techniques such as the retention of particular trees and habitats in order to mimic natural forests, they also use controlled burning to mimic natural fire disturbance regimes. Many forest certification standards require such forest management methods to be applied[16].

Box 6. Pro-Silva guidelines for close-to-nature forestry (production of timber) in central Europe[17]

- Continuous forest cover to protect soil productivity;
- Full use of natural dynamic forest processes;

[15] Bundesministerium für Ernährung, Landwirtschaft und Forsten, 2000: "Nationales Forstprogramm Deutschland

[16] see: The Swedish FSC Working Group 1998: "Swedish FSC Standard for Forest Certification, May 5, 1998, English version, http://www.fsc-sweden.org/gron/Swedish%20FSC-standard1.html

[17] See: http://ourworld.compuserve.com/homepages/J_Kuper/prosilva.htm

- Adding value by selective felling and tending at all stages of development;
- Maintaining growing stock at an optimal level;
- Working towards a balance between increment and harvesting in each management unit (i.e. in each compartment);
- Increasing forest stability, and consequently reduce production risks, through stabilisation of single trees and groups of trees;
- Paying attention to the function of every single tree in tending and harvesting;
- Avoidance of clear-cuts and other methods which destroy forest conditions;
- Abolition of rotation age as the instrument for determining when a tree should be cut;
- Undertaking renewal of the forest as an integral part of forest tending.
- Spontaneous forest renewal and forest development, through single tree harvesting and group harvesting with long regeneration periods, involving:
  - use of natural regeneration,
  - use of natural stem number reduction;
- Harvesting methods which do not harm the soil or the stand;
- Use of appropriate machines, which suit the structure and features of the forest;
- Minimise the use of additional materials (fertilisers, plant protection materials);
- Restoration of densities of game species to levels that are in balance with the carrying capacity of the forest.

Fuelwood and charcoal collection

Excessive fuelwood collection from forests will often have negative consequences for FBD for several reasons. In areas where fuelwood remains plentiful then particular preferred fuelwood species may be targeted and these can decline and eventually disappear. For instance, fire-stick harvesting from *Pinus merkusii* in north-east Thailand, a small-scale but highly destructive practice, is likely to lead to the disappearance of pine and mixed pine-dry evergreen forest associations in the region. In areas where fuelwood is in high demand, especially around major population centres, then forest communities may be degraded through excessive cutting and/or removal of biomass and nutrients. For example, extensive removal of fallen leaves for fuel in *Eucalyptus* and *Casuarina* plantations can break the nutrient cycle, lower productivity and lead to soil erosion. Specific fuelwood plantations can take the pressure off native forests to provide this commodity, while involving local villagers in thinning and pruning operations in existing plantations (in which the wood is salvaged for fuel). [Need references and would be much better to have an example from native forests than from eucalypts and casuarinas]

Harvesting of non-timber forest products[18]

Harvesting non-timber forest products (NTFPs) can strengthen sustainable management and conservation of forest biological diversity through the provision of direct benefits to people. There are exceptions however, e.g. resin tapping of important commercial timber species, including many dipterocarps in South east-Asia, may considerably weaken or kill trees if practised intensively over a long period of time and accordingly resin tapping has been banned in some countries. In Central Africa, several localised high valuable timber species, *Baillonella toxisperma*, *Milicia excelsa* and *Pterocarpus soyauxii*, are also important sources of valuable and irreplaceable NTFPs. These species may be heavily depleted by logging and accordingly there is a high conflict between their utilisation for timber and their potential to provide NTFPs (Laird, 1999). Furthermore, extraction of NTFPs can have major ecological impacts,

[18] As requested by COP, in its decision V/4, paragraph 14, the issue of impact of, and proposed sustainable practices for, the harvesting of non-timber forest products, including bush meat and living botanical resources will be addressed in-depth in a separate document for SBSTTA 7.

including adverse consequences for forest biological diversity. Such situations usually involve commercial extraction, including export to areas remote from the production area, rather than subsistence/traditional uses. They may also be associated with identification of new medicines, e.g. taxol and taxine from *Taxus baccata*, Himalayan yew (Sharma, 1999), or new markets for traditional NTFPs, e.g. yohimbine from *Pausinystalia johimbe*, yohimbe (Sunderland *et al.*, 1999), or with increasing pressure on a diminishing resource base, e.g. *Securidaca longipedunculata* ('mother-of-medicine') in Niger and *Alphitonia zizyphoides* (toi) and *Tarenna sambucina* (manonu) in Tongatapu, Tonga. *Prunus africana* is already over exploited in parts of its natural range for its medicinal bark (Dawson *et* al., 2000). Reductions in forest area and associated reductions in availability and access to traditional NTFPs, coupled with increased human pressure, are factors often associated with unsustainable extraction rates for NTFPs.

Genetic impacts of harvesting of NTFPs can be quite considerable (see: Namkoong *et al.*, 1997; Peters, 1996; Boyle, 1999). In cases where whole plants are harvested, the effects of reduced population size may be genetically significant. For a small number of particularly valuable species, entire populations have already been lost or severely depleted through over-exploitation of NTFPs. Examples include aquaje palm (*Mauritia flexuosa)* in Peru (Vasquez and Gentry, 1989), *Aquillaria* (agarwood) species in South and South-east Asia, sandalwood species[19] in the south-west Pacific (Corrigan *et al.*, 1999) and rattan species in parts of South-east Asia (Dransfield, 1989). Large-scale harvesting of the reproductive structures of plants (flowers, fruits, nuts and seeds), will directly reduce the effective size of the pool of reproductive parents and reduce genetic diversity in subsequent generations. Selective commercial harvesting of these parts can also adversely affect the genetic composition of the tree species and populations being utilised (Peters, 1990, 1996). In such cases, harvesting the 'better' fruit may remove particular, advantageous genotypes and may result in a population dominated by trees of marginal economic value with much less value as a genetic resource. In cases where a high proportion of flowers and/or fruits of a particular species is harvested on a regular basis, the most important long-term ecological and genetic impact will be a reduction in seedling regeneration and recruitment, possibly leading to eventual extinction of the population.

One approach to management and utilisation of NTFPs in the Brazilian Amazonia has been the establishment of specific "extractive reserves"(see e.g. Kageyama, 1991). Such reserves are managed by the local communities that depend on the forest for a significant component of their livelihood. It was anticipated that their establishment would result in sustainable management of the natural resources in the reserve, e.g. rubber (*Hevea brasiliensis*) or Brazil nuts (*Bertholletia excelsa*), and concomitant conservation of forest genetic resources and biological diversity. However, such export commodities are very susceptible to external market forces. Therefore, when the price of natural rubber was more than halved in the 1990s, production from Amazon forests collapsed (Assies, 1997). In more recent times, competition from cheaper Bolivian rain forest sources has contributed to a sharp decline in the export of shelled Brazil nuts from Brazil (Assies, 1999). Peters (1992) has suggested that where market-orientated extraction of NTFPs is the objective, then it would be best to focus on tropical forests dominated by only one or two useful species (oligarchic forests), rather than species-rich ecosystems. Peters (1992) gives a number of examples, mainly palm species, from Amazonia.

Overexploitation

Overexploitation can be the main factor causing extinction in the case of a relatively small number of species, principally certain valuable larger animals and tree or other plant species. While numerically small, such losses are of great concern, because they may be irreplaceable species at/or near the top of the food chain and/or keystone species and also because they often represent the loss of a component of biodiversity that is more highly valued by humans. Overexploitation typically involves the hunting of larger mammals for subsistence and sale in local markets and, more destructively, for long-distance trade

[19] One of the documented species extinctions: *Santalum fernandezianum* from Juan Fernandez islands. [reference].

and export. Bush meat hunting and trade has in many areas lead to the phenomenon of "empty forests", i.e. the total extinction of wildlife, even in natural forests (Carey, 1999) so that in many tropical forest areas, bush meat hunting has become the single most important conservation problem (see Chapter IV, Box 9.).

The Convention on International Trade in Endangered Species of Fauna and Flora (CITES) has contributed substantially to the reduction in across border trade of endangered animal and plant species, but tougher penalties and more coordinated enforcement may be needed to reduce black markets in the products of some highly-valued species. In order for CITES to be fully effective it needs to be appropriately complemented by well-implemented domestic regulations in the countries where the threatened species occur naturally.

Introduction of invasive alien species and genotypes

The ever-accelerating rate of biotic invasion of alien species and genotypes is a major element of human-induced global change. There are well-documented and numerous examples of invasive alien species and diseases causing tremendous damage to FBD. [give examples and references; e.g. elm dsease]. The damaging species may come from any taxanomic group (i.e. from micro-organisms to algae to vertebrates), as well as being from all function levels, (for instance, predators, herbivores, primary producers, parasites). Island forest ecosystems and species appear particularly vulnerable, e.g. the brown tree snake (*boiga irregularis*) introduced into Guam, in the northern Pacific, has devastated the island's birds [reference needed].

Invasive alien plant species, in particular, pose a well-confirmed and increasing danger to ecosystem integrity of many forest ecosystems. For example in the Pacific Islands there are a number of weedy trees which are having a major negative impact on FBD [reference needed for the a-d below?]:

(a)*Leucaena leucocephala* has taken over vast expanses in drier forest associations in the Pacific. It was deliberately broadcast on several war-disturbed islands after World War II (e.g. Guam) and more recently has been introduced for agro-forestry plantings.

(b)*Miconia calvescens* (purple plague), a small tree from tropical America, is the biggest invasive species problem in French Polynesia. It now covers 60% of Tahiti where is forms monospecific stands and totally shades out native vegetation. It has become established on other Pacific islands, including the biodiverse island groups of Hawaii and New Caledonia.

(c)*Spathodea campanulata* (African tulip tree) is a major invasive tree in Fiji, French Polynesia, Hawaii and Samoa, where it can take over any forest subject to disturbance, for example after cyclones. It was originally introduced as an ornamental tree.

(d)*Cordia alliodora* is a major invasive tree of native forest in parts of Fiji, Tonga and Vanuatu where it had been introduced from Central America for forestry trials and plantings.

iii. Infrastructure Development

Construction of new roads

The construction of new roads has a profound impact on the forest. Road building is considered to be one of the main causes of deforestation. Between 400 and 2000 ha may be deforested by each kilometre of new road built into intact primary forests (Contreras-Hermosilla, 2000). For example, the Trans-Amazonian highway opened up millions of square kilometres of previously inaccessible forest to colonisation and expansion of the cattle industry. Main arteries were soon followed by secondary roads that penetrated deeper into the forest, eventually producing a wide swath of deforested land on either side of the road [reference needed?]. All roads that are constructed with the purpose of providing better access to less developed regions within a country tend to push up real estate values for non-forest uses and encourage land speculation and deforestation. Logging roads are among the most important types of access roads that facilitate deforestation (Roper and Roberts, 1999) [20]

Hydroelectric development

Hydroelectric development is another important factor in deforestation. Reservoirs flood forest lands and transmission line rights-of-ways are cut out of the forest to carry the energy to consumers, causing permanent loss of forest cover. (Roper and Roberts, 1999).

Residential and cities development

Forests are also encroached upon by industrial and residential development as populations grow and cities extend outward (Roper and Roberts, 1999).

Mining and oil exploitation

Mining and oil exploration can be an important local factor in forest decline. Large mines, such as those of Carajás in Brazil and the Copperbelt of Zambia, consumed vast quantities of indigenous woodlands to supply fuel to their smelting operations before plantations of fast-growing species were established. The effects of gold mining has been widely publicised, particularly placer mining in the Amazon, but its negative impacts have affected the indigenous peoples and the quality of the water more than the adjacent forests. Oil exploration activities, such as the clearing of the seismic lines in the forests of eastern Ecuador, not only destroy the forests but also open them up to colonisation by subsistence farmers who follow the exploration crews (Roper and Roberts, 1999). Furthermore, oil exploitation often leads to severe pollution due to the leakage of pipelines; for example, several areas of the Siberian forests have been affected [reference needed?].

[20] A recent study (Laurence *et al*., 2001) reveals that the dozens of major new highways and infrastructure projects, in which the Brazilian government plans to invest $40 billion from 2000 to 2007, would lead to the destruction or heavy damage of 28-42% of the Amazon forest. The Brazilian government estimates that a maximum of 25% of the forest would be destroyed or damaged.

Box 5. Extinctions and forest fragmentation

Gilpin & Soule (1986) have grouped extinctions into two kinds: deterministic and stochastic. "Deterministic extinctions are those that result from some inexorable change or force from which there is no hope to escape. A deterministic extinction occurs when something essential is removed, or when something lethal is introduced. Stochastic extinctions are those that result from normal, random changes or environmental perturbations. Usually such perturbations thin a population but do not destroy it. Once thinned, however, the population is at an increased risk from the same or from a different kind of random event."

Aside from outright loss of forest habitat and degradation, forest fragmentation looms as one of the major threats to forest biodiversity. According to the theory of island biogeography (MacArthur and Wilson, 1967) when an area loses a large proportion of its original habitat, especially if the remnants are in fragmented patches, then it will eventually lose some of its species. Species/area curves offer a quantified prediction that the larger the forest area, the greater the number of species that can occupy the forest, and *vice versa*, e.g. reducing a habitat to 10% of its original size may lead to the loss of about 50% of species. Fragmentation exposes populations to the dangers of demographic or environmental stochasticity. Many populations are committed to extinction in the long term due to this stochasticity, leading to the so-called extiction debt (Hanski, 2000)

However, there is little direct evidence of species extinction in continental forest ecosystems. As an example, the Atlantic forest ecosystem in Brazil has been reduced to around 10% of its original extent over the past century. The theory of island biogeography predicts that around 50% of the species of this ecosystem would have become extinct as a result. A thorough survey of this area has shown that only a handful of species appear to have become extinct, although many are reduced to small populations and are therefore most likely ‘committed’ to extinction (Brown and Brown, 1992?). Another study, though, (Brooks *et al.*, 1999) points to an extinction lag time, e.g. between deforestation and species extinction. Evidence from studies on saproxylic beetles, which are a good indicator group of old-growth boreal forests, suggests that most extinctions have happened in areas with the longest history of forest use and smallest proportion of recent old-growth forests (Hanski, 2000). It is evident that the extinction is a long-term process.

Species which have the biological attributes that enable them to adapt to modified remnant forest fragments, including forest/cleared land ecotone, may increase in frequency in the new, more fragmented and disturbed forest regime. They include pioneer and secondary tree species, those with long distance seed and/or pollen dispersal, species which can better tolerate some level of inbreeding, or those which can coppice. Thus the incidence of ‘weedy’ species may increase with fragmentation.

Whilst at the broad scale, fragmentation is relatively easily mapped with the aid of aerial photography or satellite imagery, assessment of the long-term impact of such fragmentation on genetic resources of remnant stands is much more problematic. Fragmentation is a very complex threat the impact of which will depend upon many interacting factors including size, shape and location of remnant stands, and extent of gene flow between fragments, which will be affected by the degree of geographic isolation and the presence of forest corridors or scattered trees that act as bridges ('stepping stones') for movement and dispersal of species.

For tree species, the study of key evolutionary processes in fragmented populations will help in assessing the likely long-term impacts of fragmentation in given areas for given species. This will involve study of, for instance, breeding systems and gene flow through using isozymes and other molecular markers (microsatellites and RFLPs), Studies on the levels of seed set, seed viability and seedling vigour in fragmented stands may be used to provide early warning of inbreeding. Generally we know that homozygosity increases with fragmentation but that with time this trend reverses to a more heterozygous condition.

iv. Natural Hazards

Natural hazards, such as storms and hurricane damage, forest fires, floods and pests are natural disturbance regimes in forests. They can often have a positive impact on biological diversity. These disturbances, on a small or large scale, can create specific habitats that are important for the survival of a plethora of flora and fauna; they should, therefore, be mimicked or maintained in forest management (Angelstam, 1998). However, many human induced activities exacerbate these disturbances in a way that makes them an increasing threat to forest biodiversity.

Climate Change

Global climate change represents a particularly disturbing threat to FBD for several reasons. Firstly, because its impact will potentially be felt in most forest areas. Secondly, the nature and scale of its impact will be complex and require a comprehensive and co-ordinated response at both the regional level and globally. Climate change can affect biodiversity either directly through altering physiological responses or indirectly through altering interspecific relationships (Peters and Lovejoy, 1992, Kirschbaum *et al.*, 1996). The capacity of forest associations and individual component species/populations to adapt to changed climatic conditions has been greatly diminished by fragmentation, with reduced gene flow and migration options. Climate change in combination with increasing fragmentation of forests is likely to cause extinction of many species (Reid, 1992, Botkin and Nisbet, 1992, Kirschbaum *et al.*, 1996). More mobile, widespread, genetically variable species with short generation times will be best able to adapt and survive accelerated climate change. Forest tree species with restricted distributions, especially slow-growing, late successional species or those with restricted seed dispersal are especially vulnerable to climate change (Kirschbaum *et al.*, 1996). It is worth noting that any form of new disturbance or environmental perturbation is likely to have major adverse effects on forests which are rich in restricted-range endemic species (Lovett *et al.*, 2000).

Forest Fires

Lack of fire in habitats where fire is part of the ecological process of regeneration (e.g. savannah woodlands or boreal forests) can have a deleterious effect on biological diversity and its processes in the longer term. However, extreme climatic events generating fire can have devastating impacts on FBD. For example, a prolonged or abnormally severe drought can be followed by uncontrolled fire, which can destroy sensitive forest communities and species. In recent years (1997, 1998 and 2000) forest fires have been particularly severe and very widespread (in, for instance, Australia, Brazil, Central America, Colombia, Indonesia, Kenya, Mexico, Mongolia, Papua New Guinea, Peru, Russia, Rwanda, Spain, USA and western Canada). Fires devastated large forest areas that normally do not get burnt. Such unprecedented frequency and unusual occurrences of fires may be attributed to climate change. Fragmentation may prevent or inhibit recolonisation of burnt forest patches by fire-sensitive animal and plant species, thereby aggravating the negative impacts of increased fire frequency and intensity on FBD. In Samoa, two severe tropical cyclones in the early 1990s ravaged the remaining lowland rain forests, which had been opened up to greater destruction through heavy logging. These "secondary" forests are now in a state of arrested regeneration, mostly smothered by the rampant native climber (*Merremia peltata*) and increasingly subject to periodic wildfire during El Niño drought years. This example illustrates the point that FBD is especially vulnerable to the interactions of multiple threat factors. [last example requests reference]

v. Pollution

Increasing atmospheric pollution, for example so-called acid rain, has had a major degrading impact on forests, mainly in parts of Europe and around the Great Lakes in United States and Canada, lowering biodiversity values and adversely affecting forest ecosystem functioning. Results of the monitoring of forest condition in Europe show that atmospheric nitrogen and sulphur deposition affects nutrient status and tree vitality. In 30% of the plots surveyed, the nutrient status is insufficient or unbalanced. Although sulphur deposition has decreased in the last decade, nitrogen depositions originating in emissions have increased. On about 50% of the plots, N-deposition is over the critical load of 14 kg nitrogen per ha/yr, above which adverse effects are very probable, specifically on the ground vegetation. Increased pollution has also lead to changes in soil properties, which are also likely to affect biodiversity (UN/ECE and EU 2000, Forest condition report).

D. UNDERLYING CAUSES OF LOSS OF FOREST BIODIVERSITY

The underlying causes of human impact on forest degradation and deforestation and, consequently, on biodiversity loss are both numerous and interdependent and the approaches to deal with them are country specific and will therefore vary among countries (IFF, 2000; WWF *et al.*, 2000; Contreras-Hermosilla, 2000; Mc Neely *et al.*, 1995; WRI *et al.*, 1992). Some of the proximate causes discussed in the previous section, such as climate change or agricultural development, can also act as underlying causes. Among underlying causes are broader macroeconomic, political and social causes such as population growth and density, globalisation, poverty, unsustainable production and consumption patterns, structural adjustment, political unrest and war; institutional and social weaknesses such as lack of good governance, illegal logging, lack of secure land tenure and uneven distribution of ownership, loss of cultural identity and spiritual values, lack of institutional, technical and scientific capacity, lack of information, insufficient scientific knowledge and inadequate use of local knowledge; market and economic failures such as under-valuation of FBD goods and services; policy failures such as wrong incentives and subsidies and ill-defined development programmes, ill-defined or unenforced regulatory mechanisms and lack of environmental impact assessments.

I. Broader macroeconomic, political and social causes

Population growth and density

Another billion people are likely to be added to the world population for each of the next three decades. This population increase will occur mainly in developing countries, creating a strong demand for agricultural lands, forest products and "forest crops" (cocoa, coffee, bananas, etc.). To meet the associated food demand, crop yields will need to increase consistently, by over 2% every year through this period (Walker and Steffen, 1997). While possible responses to the food supply issue may be improvements in technology, better distribution of food purchasing possibilities, better nutritional education and health care, it is likely that the most immediate response will be converting more forest ecosystems to agricultural land. However, it is important to mention that the link between forest decline and population pressure remains unclear due to the complexity of the factors involved (see example in Box 7). Most studies indicate a positive relationship between population and deforestation, but most analysts are also very careful to indicate that there are other factors that obscure this linkage. For example, many authors note that loggers first make the forests accessible and then settlers occupy lands. If this is the case, then population density is the result of logging and associated initial deforestation or forest degradation, not the other way around. In addition, unless reliable information on the changes in forest cover is available, it is difficult to see the links clearly (CIFOR, 2000). At the global level, it is obvious that the enormous and still increasing demand for forest resources (timber, paper, etc) by developed countries, which do not now face population growth, is another cause of forest loss (see section on globalisation). [see comments, is this true, e.g. in the light of increasing importance of plantations?]

BOX 7. Population growth and forest cover in Indonesia

Some studies have claimed that population growth is the single most important variable explaining deforestation in Indonesia [references?] After all, the population of Indonesia has grown from about 40 million in 1900 to 200 million in 1997. The area of agricultural land in Indonesia has grown steadily in relation to this population increase, resulting in substantial loss and degradation of the original natural forest cover. However, the population-centred explanation is distorted and misleading because it rests on a flawed and incomplete view of the role of population in deforestation. Population growth is best viewed as an intermediate variable affected by others, and not simply as an independent variable that acts alone in influencing the fate of forests.

A study by Sunderlin and Resosudarmo (1999) showed the complexity of the number of variables and their interactions that have lead to forest decline in Indonesia; all of them act in conjunction with population growth. They noted that (a) a sharp decline in the rate of growth of the rural population in certain provinces is not matched by an observable decline in the rate of deforestation and forest degradation; (b) people move to forest margin areas not only because of population pressure but also because of non-population push factors, such as conversion of agricultural lands and technological change in Java, and transmigration failures in the outer islands; (c) people move to forested areas not only because of push factors but also because of pull factors, such as road construction, the infrastructural benefits offered through the formal transmigration programme and certain forms of attractive rural employment; (d) pressures on forests result not just from land clearing by rural landholders but also from increasing international and per capita domestic demand for the land under forests and for forest products; and (e) there are considerable pressures on forests that result from the indirect and direct effects of plantation development, mining and the logging sector.

Source: CIFOR, 2000.

Globalisation

At present, a fifth of the world’s population uses 85% of its resources. The globalisation of trade and these demands from the developed world for paper, timber, minerals and energy provide the incentive to exploit natural resources in the developing world. The financial and political power of large companies adds dramatically to pressures in forest ecosystems that had previously been too remote to attract attention, such as some of Central African’s rain forests and the taiga in far-eastern Russia.

In addition, the global exchange economy is based on principles of comparative advantage and specialisation and has increased in both uniformity and interdependence. In forest areas, the rapid and total conversion of forests into monocultural cash crops is widespread. But when the price of palm oil, coffee or cocoa drops, the plantation cannot quickly revert to the biologically diverse forest that preceded it, even if left alone. This is particularly the case where large-scale clearing has occurred, e.g. in south Sumatran oil palm plantations.

If environmental and social externalities (costs and benefits) are not internalised then market prices do not reflect true social values, causing allocative inefficiency. Where externalities are not internalised [should be said in a simpler way, e.g. giving explanation for externalities etc. in a footnote?], the increased economic growth from liberalised trade and investment will serve only to exacerbate rather than address environmental problems, especially in those countries that depend on the export of natural resources – e.g. forest products. The liberalisation of exchange and trade policies can improve the terms for agriculture expansion and therefore promote the clearance of forest for agricultural crops. The solution is to correct market distortions through sound environmental and sustainable development policies and, in addition, measures identified to ensure conservation and sustainable use of FBD must be implemented before bilateral and multilateral trade agreements.

International trade, investment, debt and technology transfer issues foster inequity between developed and developing countries that resemble or often reinforce those found within countries. For example, most export credit agencies and investment agencies, which finance numerous development projects, are not subject to environmental or social guidelines or standards that would ensure that they don't contribute to ecologically or socially harmful projects.

Another effect of globalisation is the increasing activity of transnational logging companies. These activities result in an expansion of destructive logging operations, violation of indigenous rights and, sometimes, widespread corruption. Most of the new investment focuses on short-term activities and the economic benefits to the exporting country are usually very low. In addition, the forests are often mined rather than managed, resulting in high levels of damage and increased access to previously untouched areas (Sizer and Plouvier, 2000).

Poverty

Poverty is both a consequence and an underlying cause of forest decline. The case of Haiti is just one of many examples showing how total deforestation, followed by soil erosion has deprived rural populations of their basis for livelihood [Reference needed]. Poverty often leads to deforestation and forest degradation. Poor people are frequently forced to slash and burn or otherwise degrade forests in response to population growth, economic marginalisation and environmental degradation. However, linkages between the rural poor and the forest resources they draw upon are complex and poverty does not necessarily lead to forest decline. Many poor people are able to adopt protective mechanisms through collective action which reduces the impacts of demographic, economic and environmental changes.

Unsustainable production and consumption patterns

Agenda 21 notes that the major cause of the continued deterioration of the global environment is the unsustainable pattern of consumption and production, particularly in industrialised countries. It further notes that while consumption is very high in certain parts of the world, the basic consumer needs of a large section of humanity are not being met. Changing consumption patterns towards sustainable development will require a multi-pronged strategy focusing on meeting basic needs and improving the quality of life, while reorienting consumer demand towards sustainably produced goods and services. Per capita consumption increased as real gross domestic product (GDP) grew at 2.9% per year while population growth was 1.4% per year. A closer look at economic trends, however, shows large disparities between and within regions. As noted in the UN Human Development Report (1998) [reference needed], 20% of the world's population, in the high-income countries, account for 86 per cent of total private consumption expenditures, while the poorest 20 per cent, in low-income countries, consume a mere 1.3%. Annual consumption per capita in industrialised countries has increased steadily at about 2.3% over the past 25 years, it has increased very rapidly in East Asia at about 6.1%, and at a rising rate in South Asia at around 2.0%. On the other hand, the consumption expenditure of the average African household is 20%

less than it was 25 years ago (UN, 2001)[21]. The effects of these consumption patterns on forest biodiversity need to be analyzed further.

Inappropriate application of structural adjustment policies

Structural adjustment policies (SAPs)[22] are very often the major element for developing countries and countries in transition to develop economic growth and regain macroeconomic stability and are in many cases imposed by international financial assistance such as the International Monetary Fund and the World Bank. If governments fail to implement mutually supporting policies, SAPs can have disastrous effects on the forests as they often provide an impetus to mine natural resources in order to meet demands for foreign exchange and to service dept payments (WWF, 2000). The reduction of public expenditure, the reduction of the role of government and the promotion of privatisation can reduce the capacity of the forest authority to enforce forest protection laws and this will open the forest for quick profit seeking private companies. The liberalisation of exchange and trade policies can improve the terms for agricultural expansion and thereby promote the dearance of forests for agricultural crops. SAPs have often been designed without careful analysis of their full potential negative impact on forests or important supportive measures have not been implemented. However, the impact of SAPs are country specific, sometimes SAPs have not lead to forest decline and a general conclusion as to the effect of SAPs on forest decline cannot be made (Contreras-Hermosilla, 2000).

Political unrest and war

One of the most important waves of large-scale forest destruction in Europe, occurring from the 15th to the 17th century, was due to the need for wood for military ship building. At the same time, dwindling wood resources for the navy prompted a number of forest protection, conservation, restoration and management measures in a number of European countries that present generations still benefit from. There is clear evidence that armed conflicts or political instabilities still correlate with an accelerated rate of forest destruction. Cambodia, Congo, Indonesia, Laos, Liberia and Sierra Leone are just a few of the countries where forests are logged for quick cash needed to purchasing military weapons and where the authorities have lost control over natural resources enabling specific actors such as the army to deplete the forests, either illegally or legally. A recent report[23] commissioned by the UN Security Council on the illegal exploitation of natural resources and other forms of wealth in the Democratic Republic of Congo demonstrates that illegal logging is linked to armed conflicts and suggests concrete measures to reduce trade in so-called “conflict timber”. Forests are also being destroyed (e.g. by herbicides) in order to eradicate sheltering places for guerrilla forces. In addition, armed conflicts cause increasing pressure on non-timber forest products, particularly bush meat for food for either the armed forces or populations that have been forced to move from the conflict areas, such as in Central Africa. This is putting some already threatened species , e.g. gorillas (see Box 1 in Chapter 4), in a very dangerous situation. On the other hand, creating military security zones has in many areas left larger areas outside economic activities. In future, many of these areas may be suitable for designation as protected areas.

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[21] United Nations Commission on Sustainable Development acting as the preparatory committee for the World Summit on Sustainable Development Organizational Session New York, 30 April-2 May 2001: Report of the Secretary-General on Changing Consumption Patterns. (E/CN.17/2001/PC8)

[22] The main elements of these policies are: correction of fiscal imbalances mainly through reductions in public expenditure; reduction of the role of the state in managing the economy; promotion of privatization; removal of obstacles to international capital flows and the formation and expansion of national capital markets; liberalization of exchange policies; removal of destructive trade policies; and deregulation of labor markets (World Bank. 1990. Adjustment lending: Policies for sustainable growth. Washington, D.C.)

[23] see : Panel on the Illegal Exploitation of Natural Resources and Other Forms of Wealth in the Congo, UN Security Council Report S/2001/357

II. Institutional and social weaknesses

Lack of scientific knowledge and inadequate use of local knowledge

In many cases there is an inadequate knowledge of natural ecosystems (their components, structure and functioning). Furthermore, destruction and decline of cultures that possess a traditional understanding of nature is resulting in a permanent loss of important complementary information on ecosystem function and management. These gaps in knowledge arise from an insufficient research effort in the study and monitoring of forest ecosystems. Such research is necessary in order to improve understanding of how various components interact, to improve information on traditional use and knowledge of biodiversity and to implement appropriate changes in ecosystem use.

Lack of good governance

The lack of good governance, rampant corruption and fraud are major underlying causes of forest decline as they surround illegal logging and other forest related crimes, such as arson and poaching. Politicians and civil servants may misuse the public power entrusted to them by, for instance, sale of logging concessions for personal enrichment, by not enforcing laws and regulations and by partaking in other illegal and corrupt activities. This generally weakens the administrative apparatus, deprives the government of income, generates incentives for "cut and run" logging operations and increases investment risks, thereby reducing incentives for sustainable forest management. The consequence in terms of forest biological loss and loss of related goods and services is often dramatic.

Illegal logging

A number of recent publications have revealed the extent of the wide range of illegal activities to be one of the major causes of forest decline (see Box 8). In the 1980s, the Philippines lost about US$ 1.6 billion per year, a large share of the country's gross domestic product, to illegal logging[24]. In 1993, Malaysian log exports to Japan were underdeclared by as much as 40%. Up to one-third of the volume of timber harvested in Ghana may be illegal and observers indicate that money injected into the country as part of a SAP led to illegal practices on a massive scale (Contreras-Hermosilla, 2000). An internal report by the Cameroon Ministry of Environment and Forests (MINEF, 1999; see also FERN, 2000) provides clear evidence of large scale, illegal activities by logging companies in Cameroon. Six companies that are amongst the largest loggers of Cameroon forests are said not to respect basic requirements of sustainable forest management. For example, they do not prepare management plans and have no respect for environmental laws.

In Indonesia, illegal logging has been recognised as the most important cause of forest decline, about half to two thirds (30-50 million m^3) of wood consumed each year comes from illegal sources. It is exacerbated by bad governance and corruption, which often include the direct involvement of military, police and forestry officials (Forest Liaison Bureau, 2000). If the current rate of deforestation continues in Indonesia, the lowland forest of the Sunda Shelf, some of the richest forests on earth, will be completely degraded by 2005 on Sumatra and by 2010 in Kalimantan (Jepson *et al.*, 2001). Global Witness (1998) described the scale of corrupt forest activities in Cambodia and stated that in 1997 much of the estimated US$184 million worth of timber felled in the country went into the pockets of corrupt officials. Illegal logging could mean the complete disappearance of Cambodia's forests in only five years time. All these studies strongly suggest a close link between illegal and corrupt activities on one hand and forest decline on the other. Greenpeace launched a series of press releases that provide evidence of the import of illegally logged wood products into the United States, Japan and European countries. According to one of their studies (Greenpeace, 2000), 80% of all wood logged in the Amazon is taken illegally.

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[24] Philippines had 50% of its primary forests left in 1946, but less than 3% of primary forest, with another 15% of secondary forest, remains. Lack of good governance is one factor for the decline [reference].

The forestry sectors of tropical countries are particularly susceptible to illegal operations and corruption. There are several reasons for this:

(a) In most tropical countries, forest activities take place in remote areas, away from the press, the public and official scrutiny.

(b) Wood, particularly in tropical countries, is valuable but not inventoried. It is thus difficult to determine how much wood is illegally extracted.

(c) Frequently, officials have substantial discretionary power. High timber values and high discretionary power held by poorly paid government officials are ideal conditions for corruption (Contreras-Hermosilla 1997).

Illegal logging is not limited to tropical countries but also occurs in other countries facing political and/or economic changes such as the Russian Federation, where an unknown, but probably substantial amount of timber is illegally logged and traded and exported, mainly to Chinese and Japanese markets but also to western Europe (see, for example, FOE, 2000).

Box 8. A catalogue of illegal acts that promote deforestation and forest degradation.

Illegal logging

- Logging timber species protected by national and international law
- Contracting with local entrepreneurs to buy logs from protected areas outside the concession
- Logging outside concession boundaries
- Contracting with local forest owners to harvest on their land but then cutting trees from neighbouring public lands instead
- Logging in protected areas.
- Logging in prohibited areas such as steep slopes, riverbanks and water catchments
- Removing oversized or undersized trees
- Extracting more timber than authorised
- Logging in breach of other contractual obligations
- Obtaining timber concessions illegally

Timber smuggling

- Exporting tree species banned under international laws such as the Convention on International Trade in Endangered Species of Fauna and Flora (CITES)
- Exporting illegal logs in contravention of national bans
- Exporting forest products in greater quantities than declared

Undergrading, undermeasuring and undervaluing timber and misclassifying species

- Avoiding royalties and duties by declaring lower value and volume of timber than is actually extracted from timber concessions
- Declaring exports of lower-priced species instead of the actual higher-priced woods
- Overvaluing services provided by overseas businesses (sometimes subsidiaries) to artificially reduce profits in the exporting country and to avoid corporate taxes

Source: Contreras-Hermosilla, 2000, based on Environmental Investigation Agency, 1996

Lack of secure land tenure and land rights and uneven distribution of ownership

The lack of secure land tenure and the inadequate recognition of the rights and needs of forest-dependent indigenous and local communities have also been recognised as major underlying causes of forest decline (IFF, 2000). Weak property rights reduce the incentive for sustainably managing the forests and unsecured land tenure is often directly related to deforestation. Local communities and indigenous people have, in many cases, traditional ways of sustainably managing the forests, ensuring that they remain viable for use by future generations. Increasing inequality of land ownership often leads to the breakdown of such common property management schemes. The rapid depletion of species and destruction of habitats occur in many countries where a minority of the population may own or control most of the land. Quick profits from excessive logging can flow to a small group of people, while the forest dependent local communities pay the price. Clear ownership rights are one of the prerequisites for developing sustainable management plans and applying regulations for ensuring the conservation and sustainable management of forests. Forest land often has a smaller value than agricultural land and, in the absence of laws that forbid deforestation, it is, therefore, cleared following privatisation. On the other hand, privatisation can be a prerequisite for ensuring sufficient investments in order to ensure the sustainable management of the forest.

Loss of cultural identity and spiritual values

As cultural homogenisation sweeps across the world, the vast range of human knowledge, skills, beliefs and responses to biological diversity is eroded, leading to great impoverishment in the fund of human intellectual resources. Loss of cultural diversity, as a result of globalisation, leads to loss of biological diversity by diminishing the variety of approaches to the coexistence of humans, other animals and plants that have been successful in the past. Loss of the different cultures also reduces the possibility of imaginative new approaches being developed in the future.

Lack of capacity, technical and financial resources

Despite all the efforts of donors and international organisations to provide money and the technology necessary to help conserve and sustainably manage forests, the lack of technical expertise and financial resources is still a very important cause of forest decline. Understaffed forest authorities, lack of knowledge about forest biological diversity and related goods and services and the lack of available qualified personal lead to little or no application or enforcement of forestry laws. In Gabon, for example, only 100 agents were available to monitor and inspect 332 logging concessions covering 86,000 km^2 (Global Forest Watch, 2000). Another underlying cause for poor forest management is the lack of appropriate forest management plans and of their implementation. Again in Gabon, only five of 200 logging companies have initiated work on a management plan (Global Forest Watch, 2000).

Deficiencies in the flow of information to decision-makers and to local communities

Where scientific or traditional knowledge exists, it does not necessarily flow efficiently to decision-makers, who may in consequence often fail to develop policies that reflect the full values of biodiversity. Information also fails to flow efficiently between central decision-makers and local communities. To complicate things further, there is a strong public reluctance to accept policies that reduce excessive resource consumption, no matter how logical or necessary such policies may be.

III. Market and economic failures

Under-valuation of forest biological diversity, goods and services

Many forest products are consumed directly and never enter markets. For instance, sawn timber, pulpwood, rattan and gums may be marketed, while food, fuelwood and medicinal plants harvested by local people will usually be consumed directly by them. Biodiversity benefits are in large part “public goods” that no single owner can claim. The benefits of biodiversity are so diffuse that no market incentives for biodiversity conservation ever develop, which ‘justifies’ government policies that further encourage conversion of the forest to other use with greater direct market values. Thus biodiversity will probably continue to decline while it remains undervalued. A challenge is to develop a ready means of attaching greater value to it in order to provide an incentive for sustainable management.

IV. Policy failures

Ill-defined or regulatory mechanisms and lack of law enforcement.

In some countries, the rise of corporate power has gone hand in hand with a breakdown in the rule of law. Economic hardship and a growing underclass have combined to create a rapid increase in illegal activity, including illegal logging, animal poaching and illegal trade (Taiga News, 1999). Lack of law enforcement is also linked to the lack of adequate financial resources allocated to the implementation of the regulations. Many national laws are too weak to provide adequate controls and when this is not the case, governments are often too weak to implement these (see Table 10). Property rights are more likely to be granted to those who clear the forests or live in cities than to forest dwellers living by the sustainable harvest of natural products. This favours the extraction of marketable products (e.g. timber) over the sustainable harvesting of products with a limited market value. The range of ill-defined regulations can cover all aspects of the causes of forest decline. As an example, in some countries there are government guidelines used to promote forest management activities that are detrimental to forest biodiversity. For instance, regulations of the former Latvian government for the management of cultivated forest areas required that every piece of dead wood be removed.

TABLE 10. Examples of policy failures that may lead to forest decline (CIFOR, 2000).

Direct government investment in the forest sector or in related sectors	• Road construction • Hydropower investments
Government command and control regulations	• Conservation area protection • Obligation to replant harvested areas • Prohibition to harvest without a permit • Obligation to prepare forest management plans as condition for intervening in forest areas • Log export bans
Fiscal, price or monetary policies	• Subsidies affecting forest raw materials or other inputs • Subsidies affecting competitive uses of lands, such as cattle ranching • Plantation subsidies • Price controls • Subsidies affecting forest harvesting or manufacturing • Forest products taxes • Subsidised credit • Foreign exchange policies affecting competitive uses of lands
Provision of services	• Delimitation, demarcation and land titling • Actions to promote exports • Settlement of frontier areas

Perverse incentives and subsidies and ill-defined development programmes

Many governmental fiscal, monetary and other subsidies and incentives create driving forces for deforestation and forest degradation. For example, transportation policies often promote the construction of roads, agriculture policies tend to promote the conversion of forests into agricultural land, resettlement programmes are frequently detrimental to forest areas and government subsidies promoting mining and hydrological infrastructure are often available. These government incentives are time and again supported through ill-defined development aid projects.

Furthermore, direct or indirect subsidies are given to economic forest operations that can damage biodiversity, such as the drainage of forests and the logging of old growth forests (Sizer and Plouvier, 2000). The most common and important type of subsidy in the forest sector is that implicit in the low forest charges paid by timber concessionaires. Although justified on grounds of promoting local development and employment, they can sometimes lead to a “boom-and-bust” situation with consequent excessive and wasteful forest degradation (Contreras-Hermosilla, 2000).

Lack of Environmental Impact Assessments or Strategic Environmental Assessments

Infrastructural development projects, structural adjustment programmes, development programmes and trade agreements have been identified as possible direct and underlying causes of forest biodiversity loss (see above). The problem is exacerbated by the fact that very often no Environmental Impact Assessment (EIA) or Strategic Environmental Assessment (SEA) accompanies the development of these projects. In addition, many of the EIAs and SEAs that are undertaken do not include a concrete analysis of the impact of the projects on the quality, size and management of the forests that may be affected.

IV. TRENDS IN FOREST BIOLOGICAL DIVERSITY

A. INTRODUCTION

It is not possible to address the issue of forest biodiversity (FBD) loss in a completely comprehensive and quantitative manner because of the complexity of FBD and the difficulties associated with its assessment at all levels. For example, there are problems in defining and classifying ecosystems and forest types; there are a vast number of undescribed tropical tree taxa and forest invertebrates and only a limited number of studies of genetic variation and processes in forest species. There is also a lack of the baseline data that are necessary to assess trends in forest biological diversity. Much more information is needed about biological impacts of fragmentation, time delays over which organisms react to changes in their environments and the impacts of FBD loss, especially in terms of species loss, to the maintenance of goods and services in various forest ecosystems. Nevertheless, governments, intergovernmental and non-governmental organizations (IGOs, and NGOs) have compiled an impressive array of statistics and measures of indicators and correlates of biodiversity and these various indicators help to provide a clearer picture of how forest biodiversity is changing.

B. FOREST COVER

Given that the maintenance of FBD is closely correlated with the area of forest cover, then the FAO global assessment of forest cover (FRA, 2000)[25] provides a useful general indicator of the change in FBD. The FAO figures, however, do not distinguish between primary and secondary forests (see Annex l for definitions), nor do they fully distinguish plantation forests from natural forests, and so those data provide only a conservative index of rates of forest change. However, forest extent is a basic measure of condition: if global forest cover shrinks, provision of goods and services from forest ecosystems will be reduced (Matthews *et al.*, 2000). According to the FAO global assessment of forest cover (FRA, 2001a), the overall decline in forest cover during the period 1990 to 1995 was 0.3% per annum and 0.22% per annum between 1990 and 2000. Between 1990 and 1995, the total area of forests decreased by 56.3 million hectares – the result of a loss of 65.1 million ha in developing countries and an increase of 8.8 million ha in developed countries. The highest rates of forest loss were recorded in the most biologically diverse moist and wet/dry tropical forests, e.g. western Africa (-1.0%), tropical Asia (-1.1%), Central America and Mexico (-1.2%) and tropical South America (-0.6%). Such declines in forest cover have been accompanied by the loss of particular forest associations, extinctions of plant and animal species and the loss of unique, locally adapted populations. The disappearance or gross reduction in given forest types and ecosystems, or the extinction of a single tropical tree species is likely to be accompanied by the loss

[25] Forest Resources Assessment (FRA, 2000), managed by FAO Forestry Department, reports information on the extent and condition of forests for the entire world using several sources, including existing forest inventory data. Inventory data are being standardized to a common classification system and reference year, viz. 2000, in order to make the figures comparable between countries. This work was carried out in close co-operation with the respective countries. As part of FRA 2000, FAO and appropriate regional institutions also used satellite remote sensing to study changes in forest cover. By interpreting a global, multi-date, objectively selected sample of about 10,000 satellite images conclusions were drawn as to the type and degree of changes in the world's forest cover over the last two decades. These studies constitute the primary source of information on the rate of deforestation, forest fragmentation and land degradation, including statistically significant information at the subregional (ecological zone) level, and provide insights into the causes of forest loss. The results of FRA 2000, including country profiles, synthesis reports and global maps, is available on the FAO Web Site and as printed reports (in press). Country profiles provide a comprehensive presentation of forest resources, including a general description of geography and the ecological setting; forest status in terms of coverage, volume and biomass, protection status and other parameters; an assessment of trends; and the sources and baseline data used. FRA 2000 reports synthesize regional and global overviews of forest status, including results from the remote sensing survey and the special studies. New global maps, with a resolution of 1 km, are presented for show forest cover, ecological zones and deforestation risks.

of several to many obligatory associated species of arthropods, especially beetles, and microflora (Gentry 1992). Depending on the methods employed and associated assumptions and extrapolations, the estimated global rates of species loss in all groups range from 1 to 9% per decade (Raven, 1988; Wilson, 1988; Reid, 1992). During the 20th Century, many regions suffered major deforestation and irreversible loss of FBD, e.g. 95% of tropical, dry forests in Central America have been converted to agriculture, often to support cattle production (Janzen, 1988).

The rate of forest decline, particularly in the past two decades, is alarming. Although there are indications that the rate of deforestation in developing countries is marginally decreasing, this trend probably does not suggest any greater emphasis towards conservation. Between 1990-1995, the annual estimated loss of forest cover (deforestation plus exotic plantation) of 13.7 million ha per year was slightly down from the previous decade (1980-1990) when annual estimated loss of forest cover was 15.5 million ha per year. However, the average rate reported for the 1990s was more than 16 million ha/year (FAO, 2001a), of which almost 95% was in the tropics, indicating that the rate has increased in the past five years. The rate of deforestation varies considerably among regions: for example rapid clearing of lowland forests in Sumatra is likely to lead to almost complete removal of primary forests within the next 5 years Jepson et al. 2001). Between 1990 and 2000, the area of global forests decreased by nearly 94 million ha, (-0.78% annually in Africa, -0.07% in Asia, -0.18% in Oceania, +0.08% in Europe, -0.12% in North and Central America and –0.41% in South America (FAO, 2001b). The overall picture is one in which deforestation is proceeding at excessively high levels that will continue to result in major losses of FBD.

C. FOREST QUALITY

Forest cover is a good general surrogate for FBD. However, it does not indicate structural changes in forest stands and ecosystems or changes in the plant species assemblages including, for example, plantations of introduced species. There is a need for research to expand the application of remote sensing methods to assess changes in tree species composition (at least change in forest types), stand structure, age classes, etc. The recent TBFRA (Temperate and Boreal Forest Resources Assessment) report (UN-ECE and FAO, 2000) includes traditional data on area of forest and other wooded areas and wood supply, as well as data on carbon sequestration, biological diversity and environmental protection, forest condition and damage and protective and socio-economic functions. This kind of additional information provides improved potential to assess changes in forest ecosystems.

The area of forest plantations increased by an average of 3.1 million ha per year during the 1990s, with half of this increase (1.6 million ha) resulting from afforestation, whereas the other half (1.5 million ha) resulted from conversion of natural forest (FAO, 2001a). Global planted forest area has been estimated to be almost 187 million ha (FAO, 2000). During 1990-2000, Asia contained 62 per cent of total plantation area, with Europe 17 per cent and North and Central America at 9 per cent. The quality of plantation forests in terms of maintaining biodiversity is often much reduced compared to primary forests, especially in the tropics where natural forests are converted to rapidly growing commercial trees, which are often exotic species (e.g. LaMonthe 1980, Estades and Temple 1999).

According to the TBFRA report (UN/ECE and FAO, 2000), 55% of forests in the temperate and boreal regions combined was recognized as largely primary (natural in the report and undisturbed by man), 41% as secondary (termed semi-natural in that report) and 4% as plantations. Figures for other wooded lands, such as tundra woodland, shrublands and savannahs were 39%, and 61%, primary and secondary, respectively. In both categories the high percentage values of natural areas were due to the vast boreal areas of the Russian Federation and Canada. However, the high values are due to the ways in which these two countries distinguish between natural and disturbed areas. If relative naturalness is analysed outside the Russian Federation and Canada, the figure for primary forests “undisturbed by man” drops to just 7% of the total boreal and temperate forest area.

Bryant *et al.* (1997) reported that 48% of the world's remaining primary forests are boreal, 44% tropical, and only 3% are temperate; (5% contain both temperate and either boreal or tropical forest). Due to their favourable climate, fertile soils and good access, temperate forests were the first to be cleared by humans and the remaining primary stands represent areas that should be fully protected.

In tropical regions, a considerable portion of existing forests can still be classified as primary, although there are great regional differences (Bryant *et al.*, 1997). In Asia and Africa, though roughly a third of the original forest cover remains, less than 10% of this original cover still qualifies as primary forest. Bryant *et al.* (1997) also estimated that outside of boreal forests, about 75 percent of the world's primary forest is threatened. Matthews *et al.* (2000) analysed the world's forests by using three forms of human activities that are known to be good indicators of environmental change: the spread of 'transition zones' (agriculture practiced at the margin of intact forest), road construction and the use of fire to clear forests. Their results were similar to those of the earlier studies of harvested forests, showing increased rates of harvest in both in primary forests and in secondary forests. There is a slow but steady increase in the amount of harvesting operations occurring in secondary forests (FAO 1995). This suggests large-scale forest quality changes in all kinds of forests, and consequently fewer forest stands regenerating naturally after cutting activities and other human caused disturbances, as well as fewer stands attaining the oldest age classes.

Use of, and change in, forest resources is a serious concern to maintaining biological diversity and proper functioning of forest ecosystems. Human disturbance of forest habitats through logging and road access can affect species diversity through habitat loss, fragmentation effects and hunting, unless a sound sustainable use programme is in place. Habitat loss is one of the three most important causes of species extinctions locally and globally in forest habitats (WCMC, 1992). There is a strong relationship between the extent of deforestation and the level of species extinctions and endangerment (Pimm et al. 1995). Skole and Tucker (1993) suggested severe deforestation in several areas of the world over the next 20 years, such as in the Amazon Basin where more than 40% of the forest will be cleared for development. This rate of deforestation will clearly have far-reaching effects on biological diversity. Even in supposedly well-managed forests, where although extinctions are few, most evidence to date suggests that there is a lack of convergence between original animal and plant communities and those in the regenerating forests (Carleton and MacLellan, 1994; Thompson *et al.*, 1999; Lomolino and Perault, 2000).

Riitters *et al.* (2000) studied global-scale patterns of forest fragmentation based on 1 km resolution land-cover maps. When anthropogenic causes of fragmentation are considered, forests are more likely to be disturbed where the climate is hospitable, soil is productive and access is easy. Boreal countries still contain a high percentage of interior forest. However, in temperate Europe and eastern North America where the conditions are more hospitable and accessible, humans have converted large areas to non-forest uses such as agriculture. Tropical rain forests remained relatively intact until access was provided by governments or industry. In the Rondonia region of Brazil, for example, the pattern of residual forest is directly related to the road pattern (Dale and Pearson, 1997). In the Amazon basin, there are corridors of fragmented forest that follow major rivers and other access routes into larger regions of interior forests. These results clearly indicate the crucial role of road networks in both deforestation and forest quality changes.

In temperate and boreal forests, advanced silvicultural practices have often caused a general homogenization of forest stands and larger forest landscapes. In particular, the size of forest patches caused by logging is reduced in comparison to patches created by natural disturbance regimes (Perera and Baldwin, 2000). In part change to forest structure and species composition in these biomes has resulted from selective logging of tree species, thinning activities, removal of dead and decaying wood and from managing the forest stands systematically, usually in a short rotation time (Maser 1990).

D. LOSS OF SPECIES AND GENETIC DIVERSITY

Current extinction rates are much higher than the rate at which species evolve, and much higher than background rates (Pimm et al. 1995). (see Chapter 3 dealing with Threats and Causes)). For bird species, current extinction rates are estimated to be at least 1000 times higher than the background extinction rate (Pimm *et al.*, 1995). The majority of both animal and plant species going extinct are those from forest and woodland ecosystems (WCMC 1992). Brooks *et al.* (1999 and references therein) accurately predicted the number of bird extinctions that have occurred in eastern North America, a region deforested many hundreds of years ago, and in more recently deforested areas in insular South-east Asia and the Atlantic forests of South America. Pimm and Brooks (1999) have predicted that of the 1,111 threatened bird species, 50%, or about 500 species, will become extinct in the next fifty years (due to the extinction debt – see Box 5, Chapter 3).

The 2000 IUCN Red List is an inventory of the global endangered status of threatened plants and animal. It uses a set of criteria to evaluate the extinction risk of thousands of species and subspecies. These criteria are intended to be relevant to all species and all regions of the world. For the first time, the 2000 Red List combines animals and plants into a single list containing assessments of more than 18,000 species (11,000 of which are threatened).

The World Conservation Monitoring Centre/UNEP and IUCN Species Survival Commission have developed a database of rare and endangered tree. Information on individual tree species is recorded in the *Tree Conservation Database* and includes the IUCN red list category, information on distribution, uses, ecology, threats and conservation measures. Summary information on individual species is published in *The World List of Threatened Trees* (Oldfield et al. 1998). The species presently included in both publications tend to be in certain tree families and genera (e.g., conifers, palms and dipterocarps) and on certain countries and regions, e.g. Africa, reflecting the particular interests and knowledge of the contributors. Together with information in appendices from Japan, Australia and elsewhere, the total number of globally threatened[26] tree species was reported to be 8,753, equating to about 9% of the world's estimated 100,000 tree species[27]. The loss of any tree genus or species will be accompanied by the loss of a variable and unknown number of obligate-associated species (including parasites, predators, pollinators and microsymbionts) and understory plant and animal species. In the tropics, the number of such associated species may be conservatively estimated[28] to be of the order of 10 to 100 per tree species.

A few generalities relating threatened species and forests should be noted:

(a)Tropical forests are extraordinarily rich in biodiversity. Threats to tropical forests therefore tend to result in very large numbers of species becoming threatened. This is especially the case if these threats take place in one of the forest areas where there are particularly high concentrations of endemic species (sometimes referred to as 'hotspots').

(b)Certain forest species are climax species. Even relatively minor disturbances can result in some species loss. There are many species that depend, for instance, on standing dead trees or on dead wood lying on the forest floor, and both of these 'habitats' tend to be absent in highly managed forests.

[26] In order to qualify as globally threatened on the basis of population reduction, the total population of the species had to experience = 20% reduction (either observed, estimated, inferred or suspected) over the past 10 years or three generations.

[27] 9% is likely to be an underestimate given the inadequate knowledge for many tropical tree species and gaps in the WMCM database.

[28] Fröhlich and Hyde (1999) estimate the ratio of host specific-fungal species to be in the range of 26 to 33 per tropical palm species. Erwin found 1,100 species of beetle from a single tropical tree (*Luehea seemannii*) and estimated that 600 arthropod species are specific to each tropical tree species (see May, 1986). Each of the 900 *Ficus* species is pollinated by a different wasp species (Janzen, 1979).

(c)There is good evidence that, at least in certain parts of the world, extinctions of forest species take place if the habitat becomes fragmented and mixed with non-forest areas (Andren 1994). Such extinctions can occur even if the forest in the surviving fragments remains undisturbed. This phenomenon has been demonstrated in particular in the Americas. [reference needed].

(d)Economically valuable forest species are currently being subjected to particularly high harvest levels in several parts of the world (UNEP 1995). Such harvests include the well-publicised trade in bush meat, but also harvest for medicinal purposes, valuable timber and a host of other non-timber forest products. Use of forest species has generally been very difficult to manage sustainably, partly because of the inherent lack of controls in many major forest areas, a lack of knowledge of populations and partly because of the slow reproductive rates of many forest species, which means that even low levels of removal can be detrimental.

Very few high priority economically important forest tree species are under immediate threat at the species level, but in all regions many species are threatened at the population level. The 11th Session of the FAO Panel of Experts on Forest Gene Resources lists over 400 tree species as being global, regional or national priorities and points out those species in need of *in situ* conservation (FAO, 2000c). A number of regional workshops have been held which document information on the state of forest genetic resources, including, although not focussing on, threatened species and populations in the Boreal Zone (Anon, 1996b), North America (Rogers and Ledig, 1996), Europe (Turok *et al.*, 1998), Sub-Saharan Africa (Sigaud *et al.*, 1998), Eastern and Southern Africa (Sigaud and Luhanga, 2000) and the South Pacific (Pouru, 2000c). For example, a report on the state of forest genetic resources in the Sahelian and North Sudan zone of Africa shows that many important tree species and populations are subject to strong pressures and are at high risk at population level; the report identifies ten species in need of *in situ* genetic conservation measures.

In temperate zones, some tree species are threatened at the population level. This applies also to well-known and economically important tree species such as *Pinus radiata* that is threatened throughout its native range. In this case, the three mainland California populations are threatened by pitch canker disease and urbanization, while on Guadalupe Island (Mexico) the species is no longer regenerating due to browsing by goats (Spencer *et al.*, 1998).

In all areas where deforestation has been extensive, it is likely that genetically distinctive, unique populations of plant and animal species have disappeared. This process of extinction of local populations will continue over the next few decades both directly through habitat loss and as a result of the time lag associated with the process of "relaxation" in which the original number of species in the fragmented area eventually falls back to a new, lower number (Diamond, 1972). Bird species in fragmented tropical forest communities will have declined about 50% of the way toward their future equilibrium after 25-100 years in large fragments of around 1000 ha (Brooks *et al.*, 1999). Many bird species have been affected in the heavily cleared woodland communities of eastern Australia and are now declining at an alarming rate, with 20% of Australian bird species now considered threatened. [reference needed].

The most pervasive threat to birds, mammals and plants is habitat loss and degradation, affecting 89% of all threatened bird species, 83% of threatened mammals and 91% of threatened plants [WCMC 1992].The primary causes of habitat loss and degradation are agricultural activities including land clearing, extractive activities and development of infrastructure and settlements. Direct exploitation is also a principal danger, affecting 37% percent of threatened birds, 34% of threatened mammals and 8% of threatened plants (WCMC, 1999). Some large species, like great apes, faced a serious combined threat to their survivel, both by predation by humans for bush meat and by the habitat loss through logging and land use changes, see Box 9. Alien invasive species are a third major source of threat, affecting 30% of threatened birds, 15% of threatened plants and 10% of threatened mammals (Hilton-Taylor, 2000). For tree species in particular, the most frequently recorded threats consist of: felling, agriculture, expansion of settlement, grazing, burning, invasive species, forest management, local use, mining exploration and tourism/leisure (Oldfield *et al*., 1998).

Threats can be recorded at species level (the whole species is endangered) or at population level (only some populations of the species are threatened). Threats at population level can severely reduce genetic diversity and the potential for breeding and domestication, such as in the case of wild fruit trees [reference needed?]. A pervasive threat to genetically diversified local tree populations, which may have developed valuable attributes, lies in the introduction of non-local forest germplasm for forest plantations or rehabilitation. This could lead to hybridization of local and introduced trees and to a dramatic dilution of native, locally-adapted gene pools in subsequent generations (Ouedraogo and Boffa 1999).

Box 9. Threats to Great Apes

Great apes are the closest living relatives of humans. The great ape group comprises the chimpanzee *Pan troglodytes*, the bonobo or pygmy chimpanzee, *Pan paniscus*, the gorilla *Gorilla gorilla*, and the orangutan, *Pongo pygmaeus*. All of these species may be keystone species in local forests as they are predominantly fruit eaters and play an important role in seed dispersal. All the great ape species are classified as "endangered" on the 2000 IUCN Red List of Threatened Species (IUCN, 2000), and certain subspecies, notably the mountain gorilla (*Gorilla gorilla beringei*) are critically endangered. The Red List notes that under a strict interpretation of IUCN categories, the common chimpanzee may, though, be classified as vulnerable.

The chimpanzee, bonobo and gorilla inhabit equatorial Africa, while the orangutan is found in South-east Asia, on the islands of Sumatra and Borneo. All the great apes face common threats to their survival, predominantly predation by humans for bush meat and loss of habitat through logging and land use change. In equatorial Africa, political unrest over the last decade has had a severe impact on the apes there (van Krunkelsven *et al.*, 2000; Vogel, 2000), with members of the military and refugees using them as a source of meat. Although meat hunting for subsistence is widespread throughout Central and Western Africa, the loss of the local agricultural economy has meant that many people, including the indigenous inhabitants of the Lomako Reserve in the Democratic Republic of Congo, are intensifying their commercial hunting efforts (van Krunkelsven et al., 2000). There is a correlation between logging activities and the bushmeat trade because new roads open up once-remote forest areas (Bowen-Jones and Pendry, 1999). Indeed, Kemf and Wilson (1997) have suggested that the bushmeat trade may now be more of a threat to African primates than habitat loss and degradation. There is also an illegal trade in African primates as pets, notably infant bonobos (Vogel, 2000).

The dramatic decline in numbers of orangutans has received much attention in recent years. In the Leuser region in Sumatra, which contains the world's largest orangutan population, the decrease is estimated to be 45 % from around 12,000 of these apes in early 1993 to less than 5,500 currently (van Schaik *et al.*, 2001),. During 1998-9, it is estimated that losses occurred at a rate of 1,000 orangutans per year (van Schaik *et al.*, 2001). If these losses continue, it is predicted that orangutans will be extinct within the next 10 years (van Schaik *et al.*, 2001). The primary cause of the decline of the orangutan on both Sumatra and Borneo is habitat loss through logging, both legal and illegal (Robertson and van Schaik, 2001). In the last 20 years, 4 million of the 13 million ha of forest on Borneo were converted to oil palm plantations (Ferber, 2000). Ferber (2000) further reports that the forest fires on Borneo in 1997 destroyed 8 million ha of forest, though some of this had been previously logged. The immediate causes of forest loss in Sumatra have been identified as weak compliance with regulations and poor law enforcement leading to extensive illegal logging practices (Robertson and van Schaik, 2001). Additionally, orangutans are in demand as pets and deaths during smuggling operations have been reported (Kemf and Wilson, 1997).

Florida panthers (*Felis concolor coryi*) provide an example of the genetic effects that occur as a result of small population size. Fewer than 50 breeding Florida panthers exist in southern Florida, USA (Maehr 1992). Low population has resulted in a high rate of random gene fixation, causing poor sperm viability, low sperm motility, probably low juvenile survivorship and an apparently elevated susceptibility to disease (Hedrick 1995). Despite protection of its remaining habitat (representing only a small portion of its former range), the panther population has continued to decline. The underlying causes of decline of this subspecies are anthropogenic influences on habitat availability (logging, agriculture, settlement) and the direct cause of problems for recovery of the panther is the loss of genetic diversity. In this case, the population has declined well below the genetically effective breeding level.

The number of endangered species in a region is correlated to the area of habitat available, habitat quality and the history of land use (Pimm *et al. 1995)*. However, extinction of a species may be time-delayed. The number of species that is expected to eventually go extinct due to past adverse environmental changes is called the 'extinction debt' (Tilman et al*.*, 1994; Hanski 1999a). In NW European boreal forest there is a gradient in the intensity and continuation of forest use from Fennoscandia, with an historically high intensity of forest use, to north-western Russia where the forests have exploited over a shorter time and at a lower intensity (Martikainen *et al*., 1996; Martikainen, 2000). This gradient is also reflected in the number of forest species in the Red List. In Finland, Norway and Sweden, forest-dwelling species form 37-51 % of all threatened species (WWF, 2001). However, many species endangered in Fennoscandia still have viable populations in Russia. For example, most local extinctions of threatened saproxylic beetles, which are habitat specialists and good indicators of old-growth forests, have happened in southern areas with the longest history of forest use and smallest proportions of recent old-growth forests (Hanski, 2000) Intensive forestry has moved from south to north, and consequently time-delayed extinctions would also be expected to spread from south to north.

Genetic diversity is needed to ensure the adaptability of a species to changing environmental conditions, as well as its continued evolution (e.g., FAO, 1993). With very few exceptions, long-term conservation of a particular species is synonymous with maintaining genetically variable populations, comprising gene complexes co-adapted to particular environmental conditions. The primary objective of gene conservation is to create conditions that will enable and enhance future evolution of the species, with a subordinate goal being to capture genes at frequencies >0.01 and to capture existing adaptations that may occur in different populations (Eriksson *et al*., 1995). This will involve maintenance of a dynamic system in which the species may continue to evolve in response to changes in its environment. At the same time, management for conservation implies the avoidance of any rapid erosion of genetic variability. Eriksson *et al.* (1995) suggested that alleles at frequencies <0.01 have little effect on additive genetic variance and hence are of little immediate effect on adaptability. Therefore, it is not necessary to save all alleles at all loci, and because evolution over the next several generations is not dependent on rare alleles, while conserving them is desirable, it is not an immediate concern.

There appears to be no direct relationship between the diversity of species or genes in an ecosystem and its biomass, productivity or role in the biogeochemical cycle (Holdgate, 1996). Relatively low-diversity systems, e.g. native forests dominated by a few tree species and various monocultural tree plantations, may be stable over many decades. Nevertheless, the likely impact of loss of biodiversity for ecosystem function, in terms of species and within-species variation, has recently been summarized (Anon., 2000: Who is this anon.?) as follows: "*The prevailing view in ecology is that diverse ecosystems consisting of viable components are more resilient than ecosystems with few species. Large numbers of species in an ecosystem provide alternative trophic webs and ways of providing ecosystem services. The loss of a few key species in a low-diversity system may lead to the collapse of the system because of the lack of alternative trophic or nutrient pathways to support higher trophic species. Maintaining high levels of genetic diversity within species, and high levels of diversity among species, is the best first approximation to maintaining ecosystem health*". Biodiversity and the presence of greater numbers of species in key functional groups may build some redundancy into the system, but because we do not know the full potential of functional groups or species it is wise not to make assumptions about redundancy. Further, species redundancy is but one of the theories explaining biological diversity. For example, other theories suggest that no such redundancies exist and that all species relate to forest function (Ehrlich and Ehrlich 1981). Species richness and genetic information may provide a buffer for certain forest types and enable healthy ecosystems to be reconstituted under a wide variety of conditions (Holdgate, 1996), and/or increase reliability of their functioning (Naaem, 1998), especially in any rapidly changing environment, as may be the case with global climate change.

TABLE 11. : Consequences of forest biodiversity loss from the perspectives of different segments of society. (From CIFOR, 2000)

Societal Group	Implications of Continuing FBD loss
Forest-dwelling indigenous communities	• Loss of spiritual values. • Disruption of traditional structures and communities, breakdown of family values, and social hardship. • Loss of traditional knowledge of use and protection of forests in sustainable ways. • Reduced prospects for preservation of forest environmental and aesthetic functions of interest and potential benefit to society as a whole. • Loss of forest products providing food, medicine, fuel and building materials.
Forest farmers and shifting cultivators	• For shifting cultivators, an immediate opportunity to survive. • Forest degradation and declining soil fertility. • Loss of access to forest land and the possibility of food crop production and reduced possibilities for harvesting forest products, both for subsistence and income generation. • Prospects of malnutrition or starvation. • Disruption of family structures and considerable social hardship.
Poor and landless local communities living outside forests	• Decreased availability of essential fruits, fuelwood, fodder and other forest products. • Reduced agricultural productivity, through loss of the soil and water protection potential of remnant woodlands and on-farm trees and loss of shelterbelt influence leading to reduced crop yield. • Reduced income generation and possibilities to escape poverty
Urban dwellers	• In developing-country situations, reduced availability (and/or overpriced) of essential forest products such as fuelwood, charcoal, fruits, building materials and medicinal products. • Loss of the amenity and recreational values of urban forests and parks and those afforded by national forest parks and wilderness areas. • Reduced prospects for assured supplies of clean drinking water and clean air.
Commercial forest Industries and forest worker communities	• Immediate large profits. • In the long-term, loss of company business and forced closure of forest operations. • Loss of jobs for forest-dependent communities, social disruption and hardship. • Loss of income and possible negative social implications of reduced income of shareholders with significant savings invested in forest industrial company stocks

Societal Group	Implications of Continuing FBD loss
Mining, oil exploration and other industrial interests	• Improved access to potentially profitable mineral, oil or other commercially valuable products located under forests. • Increased profitability of company operations and returns to company shareholders. • Politically negative impact on company operations of criticism by environmentally concerned groups.
Environmental Advocacy groups and conservation agencies	• Loss of the essential environmental functions of forests, including biodiversity, climate regulation, preservation of water catchments and fishery values, that these groups are concerned with preserving. • Loss of cultural values and social hardship for the underprivileged communities whose welfare these groups are committed to protect. • Increased problems of environmental pollution. • Loss of those forest values that could be of vital importance and/or interest to the survival and welfare of future generations.
The global Community	• Prospects that continued forest destruction will accelerate global warming with potentially negative consequences for human welfare and survival. • Continuing biotic impoverishment of the planet, loss of genetic resources, and all that implies for sustainable food production and loss of potentially valuable medicinal and other products. • Increasing pollution and toxicity of forest soils, contributing to declining forest health.
National government planners and decision makers	• Immediate escape from political pressures when impoverished populations migrate to frontier forest areas. • Loss of a potential source of development revenues with consequences of reduced employment and opportunities, sustainable trade and economic development. • Loss of the wide range of environmental functions that forests provide in contributing to societal needs and an habitable earth. • Loss of political support in situations where forestry loss and degradation adversely affect the welfare of many citizens.

E. FORESTS CONSERVED IN PROTECTED AREAS

Protected areas (PAs) may be exposed to many threats including a lack of actual protection, insufficient control of agriculture and overgrazing, illegal logging operations, wildlife and plant poaching, encroachment by human settlements, mining and human-caused fires (Dudley and Stolton, 1999). Larger-scale influences also affect PAs, including pollution, climate change and invasive alien species (Dudley and Stolton, 1999). Often, the small size of a given PA will increase the risk to species from external threats. A high proportion (59%) of PAs are less than 1000 ha in size (FAO 2000, what publication?), and they might not have long-term viability for conservation of FBD unless they are located near to and linked with other forested protected areas or subjected to comprehensive protection and management regimes. Bruner *et al.* (2001) analysed the effectiveness of protected areas in the tropical biome and found that protected areas generally stopped land clearing, but were less effective at arresting all logging. They also indicated that the capability of a protected area to maintain its integrity and role in biodiversity conservation relates directly to the extent of management and funding.

Illegal logging operations are responsible for removing valuable tree species and causing overall impoverishment of the ecology of many protected areas (Bruner *et al.*, 2001; WWF 2000). WWF has reported evidence of illegal logging in over 70 countries, including many operations in protected areas, which often appear to be particularly targeted (Carey *et al.*, 2000). Besides these problems, recreation and tourism tend to concentrate on protected areas resulting in various forms of disturbance to these ecosystems. Serious degradation to PAs is occurring in many key forested countries, such as Peru, Indonesia and the Russian Federation. The threats are not evenly distributed around the world – Africa appears to be the worst affected region. However, damage to PAs is not confined to the poorest countries, and a recent report pointed out that only one of Canada's 39 national parks is free from ecological damage (WWF, 2000). Even isolated PAs are not immune from global threats such as climate change and tourism (WWF, July 2000).

Many of the protected areas marked on maps have never actually been implemented: a phenomenon known as "paper parks" and have been subjected to considerable extractive uses (WWF, 2000). Others have been put in place but continue to be degraded and, in some cases, destroyed, either because they have not been provided with adequate staff and resources or because they face threats beyond the capacity or the range of individual managers (Borrini-Feyerabend, 1996). Others still have failed because protected area managers, or their employers, have failed to consider human needs and aspirations and the role of human communities living in and around protected areas, or else the involvement of local communities in the designating process of these areas has been deficient (e.g. Richards, 1996)..

At the same time, conservation planning has not always been systematic with the result that many PAs are poorly sited, resource-starved or badly managed, and some new reserves do not contribute to the representation of biodiversity or take into account the concerns of indigenous peoples (E.g. Horowitz 1998). As a result, the pattern of protection is still largely uneven. In particular, some forest types remain highly under-represented in Pas (ter Steege, 1998), and too many PAs still exist as isolated islands rather than integral parts of properly designed reserve networks or managed landscapes e.g. Barnard *et al*. 1998).

F. CLIMATE CHANGE

Climate change will evidently be a major threat to the forest biodiversity in the future, although the exact effects are uncertain. Many forest ecosystems will undoubtedly be subjected to greater and more frequent disturbances, which are leading to loss of FBD. This issue has been addressed in more depth in the 6th and 7th meetings of SBSTTA (UNEP/CBD/STSTTA/6/11, UNEP/CBD/SBSTTA/6/INF/13 and UNEP/CBD/SBSTTA/7/7).

To forecast the precise impacts of climate change on FBD is difficult, because they are overshadowed by the confounding effects of human-induced modifications, especially those of dominating land use patterns (Sala *et al.*, 2000). Despite varying opinions about the nature and extent of the impact of climate change on biological diversity, there is a general agreement that biological diversity will decline worldwide under most models of climate change scenarios (e.g. Bazzaz 1998, Easterling et al., 2000). Forest fragmentation is likely to increase with global warming due to increasing seasonality, desiccation and higher incidence of forest fires (Thompson et al. 1998).

Climate modelling suggests that a warming trend will lead to large changes in many areas of the earth but especially in the northern latitudes. A warmer climate will produce changes in the boreal forest biome, where the primary productivity will increase in the north but not necessarily in the south (Beuker et al. 1996), but also result in an invasion of southern species, increasing impacts of pathogens, altered fire regimes and various natural disasters caused by episodic storm events (Shugart et al., 1992, Monserud et al., 1996). The probable destruction of permafrost accompanying climate change (Dyke and Brooks, 2001) and human land use will cause major landscape degradation and loss of biological diversity over large areas (Sala et al. 2000). Areas of northern taiga and treed tundra are predicted to be replaced by more productive boreal forests as climate warming occurs, while some drier southern boreal areas may become savannahs (Solomon 1993, Suffling, 1995; Thompson *et al.,* 1998).

The capacity of forest associations and individual component species/populations to adapt to changed climatic conditions has been greatly diminished by fragmentation, with reduced gene flow and migration options (Thompson et al. 1998). Climate change in combination with increasing fragmentation of forests is likely to cause extinction of certain species, especially among arthropods and plants (Reid, 1992; Botkin and Nisbet, 1992; Kirschbaum *et al.,* 1996), More mobile, widespread, genetically variable species with short generation times will be best able to adapt and survive accelerated climate change. Forest tree species with restricted ranges, especially slow-growing late successional species or those with restricted seed dispersal, are especially vulnerable to climate change (Kirschbaum *et al.*, 1996). Forests that are rich in restricted-range endemic species are likely to be particularly adversely affected (Lovett *et al.* 2000). Further, changes brought on by climate warming must be viewed in the context of already disturbed landscapes, making comparisons with historical post-glacial migrations problematic (Kuuluvainen et al., 1996).

Natural fires are a crucial element for the succession of many forests, especially in boreal areas. Prescribed burning, mimicking wildfires, should be used to a greater extent in restoration of forests in conservation areas and also in some managed forests. In the changing climate, however, natural and human-caused fires can have deleterious impacts on FBD; for instance, after the predicted prolonged periods of drought (see in more detail UNEP/CBD/SBSTTA/7/7). These fires have destroyed many important fire refugia on which many forest species intolerant to fire are dependent. Both the unusual frequency and new regional occurrence of fires may be attributed to climate change.

G. THE NEED TO DEVELOP MONITORING PROGRAMMES

Monitoring is an important component of a sustainable forest management programme (McLaren et al. 1998). Monitoring should, over time, provide information on the development of key-parameters or species that indicate important trends in forest biological diversity; this information should be used to evaluate the current management and to contribute to continuous improvement. Ordinarily, monitoring is used to measure achievement of pre-determined objectives within the context of an adaptive management programme (Walters and Holling 1992).

Scientifically based general indicators for measuring biodiversity are not yet available for all forest types or countries, although many countries have begun reporting on various indicators. Loss of (natural) forest cover is sometimes used as a coarse indicator (see Chapter I) at the global or regional level. Species or species groups can be used as indicators at the national, landscape or ecosystem levels, but very often information on species is limited and difficult to obtain. Nonetheless, species or groups of species are likely to be used for many reasons as national or local indicators (Noss and Cooperrider 1991). As an alternative, structural characteristics (tree species composition, tree age, standing dead wood) can be important parameters on landscape or ecosystem scales and might make suitable local indicators (Spies and Franklin, 1991).

The use of advanced technology has become routine among developed countries in forest management programs, but lack of such technologies may affect the capacity to protect biodiversity, Forest types, ecosystem types and landscape structures can be monitored to determine changes in tree cover, (main) tree species composition, tree age-classes and availability of successional stages by using various remote sensing methods linked to computer imaging programs, and geographic information systems. Various sophisticated modelling tools exist to predict changes in forest ages and types based on known and expected rates of logging, autecological rules and expected silviculture applications. Modelling, based on sample data and expert opinion, has become increasingly important in forest management planning to predict large-scale changes.

Monitoring to examine the effects of forest clearing and forest management is a difficult task because of our incomplete knowledge of species in many countries, difficulties in selecting aspects to monitor, lack of classifications of local ecosystems, logistical and cost considerations, and lack of required technologies. At the species level, rare, threatened and endangered species must be monitored to track populations and indicate effects of management programmes and protection schemes. However, other species can also be used to track broader changes in forest condition. Many techniques exist for monitoring species and communities of plants and animals. Regardless of method, an important aspect is that the methods should be consistent among surveys and observers to enable comparisons among years and locations. Surveys must also be designed to allow proper data analysis, which gives valid results. Monitoring should also be conducted to test hypotheses that relate to changes (possibly specific changes) in forest capability to support biological diversity (Walters and Holling 1992).

In Canada, McLaren *et al.* (1998) suggested that a series of criteria should be applied to the selection of vertebrate indicator species based on three main categories: biological factors, available methods for censuses, and political considerations (see Annex II.). While the criteria of McLaren *et al.* (1998) were established for vertebrates, they apply equally to plants and invertebrate species. Noss (1999) has suggested a series of groups from which indicator species might be selected. These include: area-limited species which require the largest minimum dynamic areas to maintain populations; dispersal-limited species which have only limited capacity to move among patches of habitat; resource-limited species that require a resource(s) that is in short supply; process-limited species which are sensitive to certain ecological process which are infrequent or at low levels; keystone species; and species with limited geographical ranges.

Several models exist that can be used to predict total species based on the number of species detected in a survey. Rarefaction models (e.g., Krebs, 1989) have been employed for many years to suggest upper limits to species richness. Soberon and Llorente (1993) reviewed three species accumulation curves where sampling time was the independent variable: Clench equation, exponential, and logarithmic models. This approach has not received wide attention from conservation biologists, but the results from Soberon and Llorente suggest that the approach has merit in certain situations: for example, when comparing species presence among landscapes where temporal restrictions apply. Such modelling of communities offers an additional approach to ‘indicate’ and predict change in forest function.

The difficulties and shortcomings with basing biological diversity conservation efforts solely on individual species management plans have been well documented (Simberloff, 1998; Maddock and Du Plessis, 1999). Nevertheless, both individual and aggregated species data, such as the ‘hotspot’ or ‘centres of diversity’ and ‘centres of endemism’ approach (UNEP, 1995) will likely continue to play a role in conservation of forest biological diversity. Priority forested areas for conservation have sometimes been identified by comparing “biological diversity hot spot” data with forest cover data. Hot spot data sets include those developed for endemic bird species, WMCM/IUCN ‘centres of plant diversity’, WWF-US Global 2000 eco-regions, and Conservation International’s hotspots [give references for these concepts]. WWF and IUCN have identified 234 priority centres of plant diversity or endemism, each with at least 1000 vascular plant species, of which 100 or more are endemic. As mentioned previously, one of the main problems with this approach is that ‘hotspots’ vary among different taxonomic groups. Furthermore, species distribution data are very incomplete, e.g., data are only available for about 3 percent of described species (WCMC, 1990). However, ‘hotspots’ may reflect our degree of knowledge of richness in particular groups of more obvious taxa, in particular, and better-sampled areas, as much as representing real concentration of species diversity. However, if a high degree of endemism exists within a hotspot, the relative importance of the area is increased (UNEP 1995).

Monitoring and inventory of genetic diversity, the finest scale, may be appropriate for several purposes. First, in the case of endangered species it is often important to determine whether a particular population is indeed a separate species or subspecies to meet designation criteria. Second, it may become important to understand whether population genetic processes have reduced variability within a species to a point where it may be affecting breeding success. Inventorying genetic diversity might be used to determine the future usefulness of various potential crop plants.

H. CURRENT POSITIVE TRENDS IN FORESTRY AND FOREST POLICIES

Although increasing forest use has given rise to many threats to FBD, and endangerment of the forest biota has increased, there have also been some positive developments in international and national forest policies. Many intergovernmental commitments have been made to promote sustainability within the forestry sector, such as the decisions made at UNCED (Forest Principles, Agenda 21, Chapter 11 on Combating deforestation and others) and subsequently in The Intergovernmental Panel on Forests (IPF, 1995-1997) and in The Intergovernmental Forum on Forests (IFF, 1997-2000). The process will continue in the United Nations's Forum on Forests (UNFF). During this process knowledge, as well as public awareness, about sustainable forest management, has greatly increased, and the concept and principles of sustainable forest management, including protection of forested areas, have been widely introduced. For example, policy measures to implement sustainable forest management have been increasingly adopted and perverse policies have been abandoned.

An objective to forests as ecosystems has been widely adopted in the national forest policies, particularly in developed countries. New tools to implement sustainable forest management, such as criteria and indicators, are being introduced. Guidelines for good forest practices, including low impact logging, natural disturbance guidelines, and protection of species are being adopted as a part of the ecosystem management methods. At present, it is still too early to say what is the success may be of these various approaches in maintaining forest biological diversity. Conservation of biological diversity has also become a feature of national forest policy and planning in many countries. However, the level of integration of conservation and forestry issues varies greatly from one country to another.

Preparation of national forest programs (NFPs) in many countries resulted from the IPF/IFF process and the assistance provided by international organisations. NFPs encompass the full range of policies, institutions, plans and programmes to manage, utilise, protect and enhance forest resources within a given country. Under the Convention on Biological diversity (CBD) most countries have prepared country studies and reports on the state of biodiversity. A further step was preparation of national biodiversity strategies and action plans (NBDSAPs). These provide general information on diversity of forest species and ecosystems, and their principal uses in particular countries. The strength of both IPF/IFF and CBD processes is related to the political commitment generated and the flexibility that allows each country to determine its conservation and forest programmes according to national conditions. Various coordinating bodies are also established in many countries in order to enhance collaboration between various stakeholders and to integrate biodiversity issues into sectoral policies.

The concept of the use of criteria and indicators for sustainable forest management was perhaps the most comprehensive policy-related, yet practical, tool to gain popular acceptance following UNCED. This is reflected in the worldwide application of this concept, which began several years ago. Within the forest sector, the approach of identifying and developing indicators has been unique. Under regional and international processes and initiatives, the components of sustainable forest management have been characterised by criteria for which indicators are used to transform the concept into a measurable form. At present there are nine regional or international processes that have developed criteria and indicators for sustainable forest management for the specific conditions in the respective regions. The nine are: ITTO, CIFOR, ATO, Montreal-Process, Pan-European Forest Process (MCPFE), Tarapoto-process, Dry Zone Africa-process, Near East–Process and Lepaterique-Process. Some countries have already made changes to their forest laws and institutions to meet the requirements of the concept of criteria and indicators. Indeed, this concept is increasingly used in the context of national forest programmes and, more recently, it is being used at the local and forest management unit levels. In some countries and initiatives, concept using criteria and indicators has been linked to certification, which has gained increasing attention in recent years.

The role of regional initiatives in sustainable forest management has been important in implementing and operationalising international initiatives. For example, in the context of the Ministerial Conference for Protection of Forests in Europe (MCPFE) and its follow-up process, the ministers responsible for forests in Europe adopted six criteria and endorsed the associated indicators for sustainable forest management by signing Lisbon Resolution L2 in 1998[29]. Criterion 4 and the related seven quantitative and 26 descriptive indicators are focused exclusively on biological diversity. They relate to the maintenance, conservation and enhancement of biological diversity, including rare and vulnerable forest ecosystems, threatened species and biological diversity in production forests.

Awareness of the importance of forests to the approximately 400 million people that live in and around them and largely depend on them for their well-being and subsistence has been increased, and there is also a rise in willingness to accept issues related to rights, needs and participatory possibilities of indigenous people and local communities in the context of forest conservation and management. This positive development includes interests by donor institutions to collaborate directly with indigenous and local communities, policy revision by many actors (e.g. WWF principles; EU policy on cooperation, donor agencies, WB, UNDP, etc.), increased acceptance of traditional knowledge and collaborative participation in forest conservation and management, including participation of indigenous people in the management of protected areas However, these changes have happened largely at international level and have not yet materialized sufficiently in national policies.

Transfer of knowledge to efficient policies to promote sustainable development involves an array of difficult choices. For example, while we know that forest clearing for crops and pasture, uncontrolled commercial logging for timber and expanding infrastructure all contribute to deforestation and degradation, the fundamental problem facing policy-makers is how to address the underlying causes. One way to improve conservation of biodiversity while using the forests is to demonstrate, in a commercial sense that it can be done. Demonstration areas are one way to illustrate not only effective policies, but how such policies can be implemented. One such example is that of the Canadian and International Model Forests Network (see Box 10).

Box 10. The Canadian and International Model Forests Network

1. A key way to improve forest management is to demonstrate effectively that sustainable forestry can indeed be accomplished. In Canada, demonstration of sustainable forest management is conducted through a network called the *Model Forests*. This effort was initiated by the Canadian Forest Service in 1992, to address the challenge of balancing the extensive demands on forests, with the needs of future generations. The principle behind the programme is that each model forest serves as a demonstration area, with partners representing a diversity of forest values, who work together to achieve sustainable forest management. These forests act as working laboratories in which leading-edge techniques are researched, developed, applied and monitored. A model forest encompasses a area scale land base (several 1000's of km^2), where the participants have a direct interest and influence over the uses in the forest.

[29] Furthermore the Pan-European Operational Level Guidelines (PEOLG) for SFM were endorsed.

2. A model forest partnership typically includes industrial companies, parks, landowners, governments, aboriginal people, academic institutions and environmental groups. Each forest provides a forum where the partners can gain a greater understanding of conflicting views, share their knowledge and combine their expertise and resources to develop innovative, region-specific approaches to sustainable forest management. The result of this grass roots approach is solutions that work and earn local support.

3. Although research and innovation is taking place in each model forest, some activities are pursued at the national level. This allows the people involved in model forests to come together and share their unique perspectives as they work toward sustainable forest management on a national scale. In this way, the national perspective is ensured and results from these projects can be integrated into programs taking place at each site.

4. Currently, there are three national initiatives taking place within the CMFN. Each is an integral component of the Model Forest Programme and crucial to the success of the network as a whole:

   1. Achieving sustainable forest management (SFM) is a complex challenge that requires decision-makers to seek a balance between social, economic, cultural and environmental objectives for a forest area.

   2. Local level indicators (LLI) are developed to suit local and regional conditions, they provide the framework for monitoring on-the-ground changes and enable assessment of the many components of sustainable forest management.

   3. Criteria and indicators (C & I) such as the national Canadian Council of Forest Ministers (CCFM) C & I and international Montreal Process C & I allow for the measurement of progress towards sustainable forest management along national scales. As a common framework, each model forest used Canada's six criteria for sustainable forest management as defined by the CCFM: 1.conservation of biological diversity; 2. maintenance and enhancement of forest ecosystem condition and productivity; 3. conservation of soil and water resources; 4. forest ecosystem contributions to global ecological cycles; 5. multiple benefits of forests to society; 6. accepting society's responsibility for sustainable development.

5. Model forests, in collaboration with their partner organizations, are exploring ways to collect and analyse data to effectively and efficiently monitor their local level indicators. Information on various local level indicators is collected and used to report and measure on the region's progress towards forest sustainability.

6. An important component of the programme in each model forest is biodiversity research, with priorities established locally and nationally.

7. Because the Canadian Model Forest program was successful, other countries have now begun to develop their own model forests. The first country to join the International Model Forest Network was Mexico, and Model Forests in Calakmul and Chihuahua signed on as members in 1993, followed by Mariposa Monarca in 1995. Russia followed with the Gassinski Model Forest in 1994, and the United States has now designated three model forests: Cispus, Hayfork and Applegate. Other countries currently developing Model Forests include Argentina, Malaysia, China, Japan and Vietnam. Other countries expressing interest in developing model forests include

Australia, Ecuador, Indonesia, Southern African Development Community, and the United Kingdom. The International Model Forest Network represents a greater diversity of the major forest ecosystems around the world. An objective for the IMFN is to represent all forest types ranging from the boreal/subarctic coniferous regions to the tropical rain forests.

I. IMPLICATIONS OF DECLINING TRENDS IN FBD

Forest biodiversity is a broad concept with many dimensions. It includes diversity of genes, species and ecosystems, as well as that of forest landscapes. Besides producing many kinds of goods for humans, forest biodiversity has great cultural and intrinsic values. FBD also maintains important ecosystems functions and services, it provides a basis for various livelihoods and for sustainable land use and development. Diversity of genes, ecotypes and habitats is also the best insurance against adverse changes in the future, regardless of whether changes are natural or human-caused lie climate change or degradation of genetic resources. Naturally diverse forests have greatest potential to adapt to unpredicted changes, as well as to provide new goods and services.

All forest areas have some role in conservation and use of forest biological diversity, but any given forest area cannot produce all goods and services. Maintenance (and where appropriate enhancement) of FBD is an important aspect of conservation and sustainable forest management; this applies to the whole range of forests from protected primary forests, managed (semi-)natural forests, plantations and other ecosystems that include elements of FBD.

WWF and IUCN have identified five priority objectives to halt and reverse the decline in the global forest estate (WWF/IUCN, 1996; WWF 2001):

(a)To establish a network of ecologically representative, socially beneficial and effectively managed forest protected areas

(b)To achieve environmentally appropriate, socially beneficial and economically viable management of forests outside protected areas

(c)To develop and implement environmentally appropriate and socially beneficial programmes to restore deforested and degraded forest landscapes

(d)To protect forests from pollution and global warming by reducing polluting emissions and managing forests for resilience to climate change

(e)To ensure that political and commercial decisions taken in other sectors safeguard forest resources and result in a fair distribution of associated costs and benefits.

It is likely not possible to maintain all of the characteristics of a natural forest landscape and forest structure in protected areas. For this reason, the implementation of sustainable management practices is of utmost importance. Nature reserves can in addition, and when actually managed, complement biodiversity programs in managed forests. Protected areas also provide scientific ‘benchmarks’ against which to measure progress towards sustainability in forest management.

Protected forest areas are of special interest because their primary aim is the maintenance of FBD. These areas, however, should be managed in context with surrounding areas because forests are not static in time. Semi-natural managed forests are important in maintaining many elements of FBD and producing goods and services. This also concerns plantation forests, in which these diversity aspects should be enhanced. Further, protected areas should also be a part of a reserve network at a larger, regional scale.

Reserve networks should be both representative and complementary, which means that all habitat types of natural and semi-natural ecosystems should be represented in the network. Especially lowland forests on fertile soils, riverine and many coastal woodlands are underrepresented in conservation area systems of many countries.

In selecting individual reserves several aspects should be taken into account (Spellerberg, 1994), including size and extent of the area, diversity of species, communities and ecosystems, naturalness, rarity and commonness, and fragility. The larger the area the more species that are included and the larger the population size of the individual species (Connor et al. 2000). In addition, the negative impacts of events in surrounding habitats are proportionately smaller on large areas. However, many species and habitats occur in specific, small-scale sites, such as springs, small wetlands and other various sites with varying edaphic, hydrological and microclimatic conditions (Rabinovitz et al. 1986). In such cases, protection of small sites/areas is well founded. Diversity of species, communities and ecosystems is important, but species-poor habitats should be emphasized if they consist of rare or endangered species or unique habitats. Fragility of a habitat should also be taken into account in founding reserves.

In addition to foundation of forest reserves, forest management practices should take into account the demands of sustaining biodiversity. The practices are largely dependent on the forest type, its regeneration regime and land-use history. Important aspect of a more ecological forestry is the consideration of key biotypes (see Box 11), i.e. valuable, often small-scale habitats or biotopes of high natural values that should either be left untouched or managed with special care, and ecological corridors connecting high conservation value areas. These structures are important for conservation of many species, and they may ease species' dispersal within managed forest landscapes.

Forest planning should be carried out on a landscape-scale to ensure long-term sustainable use. Landscape

Box 11. Key biotope concept and forest management

One important element of more ecological forestry in the managed forests is the proper management of key biotopes, i.e. biologically important small-scale habitats. The key biotope approach has also been introduced in the Swedish and Finnish forestry legislation and practical guidance in forest management, and it has been adapted in last years also in Estonia.

The key Woodland Habitat identified in Swedish forests is an area where one or more Red-listed species occur (Skogstyrelsen, 1998). The number of woodland key habitats is estimated to be around 80,000 with an average size of 2.3 ha. The key habitats in Finnish Forest Legislation are valuable, small-scale (usually 0.2-1 ha) habitats or biotopes of high natural value. They include natural springs, brooks and small ponds with their surroundings, herb-rich woodlands, rich fens, grass- and herb-rich wooded swamps, rocky crevices and gorges, rocky cliffs and underlying herb-rich forest stands, and some sparse woodland biotopes (exposed bedrock, boulder fields, sparsely wooded mires and alluvial forests). It has been estimated that these habitats cover 0.5-1.5 per cent of the forested land, and if other important habitats – various ecotones, shore woods and disturbed key biotopes, which could be restored - are taken into account, the areas is perhaps 2-4 per cent.

The identification and management of key habitats varies at present quite a lot, but their adoption in the practical forestry guidance will give better results in the longer term. The scientifically verified knowledge about the effects of preservation and management of key biotopes on forest biota is scanty, as yet. However, they are important for species occurring in specific sites in forest landscapes, e.g. in terms of topography, edaphy or hydrology. A great deal of locally rare and threatened flora and fauna (such as vascular plants, mosses and liverworts, many groups of invertebrates) are found in these habitats.

ecological planning has been introduced in recent years in many forest countries, e.g. all state-owned forest lands in Finland (Karvonen, 2000). Such ecological planning involves an understanding of the natural disturbances, which controls the landscape structure or pattern. In Sweden, this landscape approach has been used on the large forest estates of major forestry companies. One earlier application of this thinking is the so-called ASIO model in Sweden. This model has been developed for forestry to better mimic the disturbance dynamics of different forest types (Angelstam *et al*. 1993, Angelstam 1997, 1998). The model is based on the intensity and significance of fire occurring in different types of forests. Forests are divided into whose, which burn almost never (A), seldom (S), intermediately (I) and often (O). The model suggests that different forestry practices, based on natural disturbance models, should be carried out in these four fire frequency classes.

A number of traditional and modern sustainable livelihoods exist, which can be well combined with forest protection programmes and sustainable forest management. These include the harvesting of many non-timber forest resources, like collecting food products (wild fruits, mushrooms, berries), medicinal plants, natural rubber, fibres etc. If properly planned, these will also have the dual benefit of revitalising local economies and fighting rural depopulation. Also nature-oriented tourism, if well implemented, can be an actively compatible and even supportive, of the conservation of forest protected areas. When developing these activities, needs of Indigenous Peoples and local communities should be fully taken into account. Participatory approach and community involvement are key issues in the sustainable management of natural resources.

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ANNEX I

USE OF TERMS

Forest:
The group considers the FAO definition of a forest as the basic one (FAO, 1998; FRA 2000), but acknowledge that many other useful definitions of 'forest' exist in published form. The fact that 'forest' has been defined in many ways is a reflection of the diversity of forests and forest ecosystems in the world and of the diversity of human approaches to forests. In this document, a forest is a land area of more than 0.5 ha, with a tree canopy cover of more than 10%, which is not primarily under agricultural or other specific non-forest land use. In the case of young forests or regions where tree growth is climatically suppressed, the trees should be capable of reaching a height of 5 m *in situ*, and of meeting the canopy cover requirement.

Forest biome:
A biome is the broadest forest classification unit. This reflects the ecological and physiognomic characteristics of the vegetation and broadly corresponds to climatic regions of the Earth. In this document, it is used in reference to boreal, temperate and tropical forest biomes.

Forest type:
Within biomes, a forest type is a group of forest ecosystems of generally similar composition that can be readily differentiated from other such groups by their tree and undercanopy species composition, productivity and / or crown closure.

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

Forest biological diversity:
Forest biological diversity means the variability among forest living organisms and the ecological processes of which they are part; this includes diversity in forests within species, between species and of ecosystems and landscapes.

Primary forest:
A primary forest is a forest that has never been logged and has developed following natural disturbances and under natural processes, regardless of its age. It is referred to 'direct human disturbance' as the intentional clearing of forest by any means (including fire) to manage or alter them for human use [Is this clear enough?]. Also included as primary, are forests that are used inconsequentially by indigenous and local communities living traditional lifestyles relevant for the conservation and sustainable use of biological diversity.

In much of Europe, primary forest has a different connotation and refers to an area of forest land which has probably been continuously wooded at least throughout historical times (e.g., the last thousand years). It has not been completely cleared or converted to another land use for any period of time. However traditional human disturbances such as patch felling for shifting cultivation, coppicing, burning and also, more recently, selective/partial logging may have occurred, as well as natural disturbances. The present cover is normally relatively close to the natural composition and has arisen (predominantly) through natural regeneration, but planted stands can also be found. However, the suggested definition above would include other forests, such as secondary forests.

Secondary forest:[30]
A secondary forest is a forest that has been logged and has recovered naturally or artificially. Not all secondary forests provide the same value to sustaining biological diversity, or goods and services, as did primary forest in the same location. In Europe, secondary forest is forest land where there has been a period of complete clearance by humans with or without a period of conversion to another land use. Forest cover has regenerated naturally or artificially through planting.

Old growth forest:
Old growth forest stands are stands in primary or secondary forests that have developed the structures and species normally associated with old primary forest of that type have sufficiently accumulated to act as a forest ecosystem distinct from any younger age class.

Plantation forest:
A plantation forest may be afforested land or a secondary forest established by planting or direct seeding. A gradient exists among plantation forests from even-aged, single species monocultures of exotic species with a fibre production objective to mixed species, native to the site with both fibre and biodiversity objectives. This gradient will probably also reflect the capability of the plantation forest to maintain 'normal' local biological diversity. [Possible addition: Depending on the intensity of management, species mix, presence of understorey vegetation and natural species regeneration within the planted forest, and proximity to natural forest stands or seed sources, the planted forest may support, to a lesser or greater extent, local biological diversity]

Degraded forest:
A degraded forest is a secondary forest that has lost, through human activities, the structure, function, species composition or productivity normally associated with a natural forest type expected on that site. Hence, a degraded forest delivers a reduced supply of goods and services from the given site and maintains only limited biological diversity. Biological diversity of degraded forests includes many non-tree components, which may dominate in the undercanopy vegetation.

Agro-forest:
An agro-forest is a complex of treed areas within an area that is broadly characterised as agricultural or as an agro-ecosystem.

Reforestation:
Reforestation is the re-growth of forests after a temporary (<10 years.) condition with less than 10% canopy cover due to human-induced or natural perturbations (FAO, FRA 2000).

Afforestation;
Afforestation is the conversion from other land uses into forest, or the increase of canopy cover to the 10% defined threshold for forest (FAO, FRA 2000).

Forest fragmentation:
Forest fragmentation refers to any process that results in the conversion of formerly continuous forest into patches of forest separated by non-forested lands.

Habitat loss:
Habitat loss, used with reference to an individual species, is the permanent conversion of former (forest) habitat to an area where that species can no longer exist, be it still forested or not.

Forest species:

[30] Locally in Malaysia and South-east Asia, secondary forests refer to highly degraded forests; this is not the intent of the term in this document.

A forest species is a species that forms part of a forest ecosystems or is dependent on a forest for part or all of its day-to-day living requirements or for its reproductive requirements. Therefore, an animal species may be considered a forest species even if it does not live most of its life in a forest.

Native species:
A native species is one which naturally exists at a given location or in a particular ecosystem, i.e. it has not been moved there by humans.

Endemic species
An endemic species is a native species restricted to a particular geographic region owing to factors such as isolation or in response to soil or climatic conditions.

Alien species
An alien species is a species, sub-species or member of a lower taxon that has been introduced outside its normal past and present distribution; the definition includes the gametes, seeds, eggs, propagules or any other part of such species that might survive and subsequently reproduce (GISP, 2001).

Invasive alien species
An invasive alien species is an alien species which becomes established in natural or semi-natural ecosystems or habitats. It is an agent of change and threatens native biological diversity (IUCN, 2000).

ANNEX II

CRITERIA FOR SELECTION OF VERTEBRATE INDICATOR SPECIES FOR FORESTS IN CANADA

I. Biological Selection Criteria

1. Species is dependent on forest habitats.
2. Species is responsive to forestry practices.
3. A range of body sizes/home range sizes should be represented.
4. A range of life history strategies should be represented.

4a. All trophic levels should be represented in the suite of species chosen.
4b. Year-round residents should be given priority over migrants in northern forests.
4c. Habitat specialists and generalists should be included.

5. Biologically rare species should be selected, particularly if their rarity is habitat- related.
6. Species should inhabit a range of forest habitats and structural characteristics (e.g., upper, middle, and lower canopy).
7. Any known keystone species should be selected.

II. Methods Selection Criteria

1. The species must contribute to testing a valid hypothesis with respect to factors which might influence the species
2. A sampling protocol must be available.
3. Species should be distributed so that a statistically valid sampling is possible.
4. Sampling should be cost effective.
5. Whenever possible, a control area, where populations are determined by natural factors, should be used as part of the sampling design.

III. Status Selection Criteria

1. Nationally or regionally featured species.
2. Species with a low or diminishing habitat availability at the ecoregional scale.
3. Species should be ones for which a given jurisdiction bears a high level of conservation responsibility (or endemism) based on proportion of total range within the country.
4. Species is easily recognizable by the public and politicians.

Source: McLaren *et al.* (1998)

ANNEX III

DATA MANAGEMENT AND SCIENCE RESEARCH AGENDA TO UNDERSTAND AND MANAGE FOREST BIODIVERSITY

(a) Determine relative importance of factors affecting forest biological diversity

Current forest management practices
Natural disturbances
Current and past land use practices
Alien species
Climate change
Pollution
Habitat fragmentation

(b) Develop methods to compile an inventory and to monitor biological diversity

Mapping of biodiversity associated with forests at a variety of scales
Development and testing of indicators at coarse and fine scales
Classification and assessment of forest types, ecosystems and landscapes
Predictive modelling of indicators for forest types and landscapes
Develop sustainable extractive schedules
PVA for important species

(c) Develop systems for decision making

Gap analysis
Data management, access, integration and analysis of information
Forest mapping under GIS and remote sensing
Integration of primary databases with biological databases

(d) Develop strategies for conservation of biological diversity under sustainable use of forests

Protected areas and heritage programme with objectives and maintenance strategies
Ecosystem management strategies
Adaptive management policies and procedures
Restoration of degraded forest ecosystems
Ex situ gene conservation
Adaptation of management practices to global climate change
Mechanism to set old growth and primary forest objectives
Assessment of public and indigenous peoples' acceptance of strategies
Determining centres of endemism
Determining rare and threatened species and their management
Improved understanding of traditional knowledge

(e) Research agenda to support management options

Understanding processes influencing biodiversity at a range of different scales
Relationships between biological diversity and productivity, biological diversity and stability and biological diversity and function within forest subtypes or ecosystems
Proper adaptive management experiments

Effects of forest use on species, communities, ecosystems and landscapes functioning
Testing umbrella properties of indicators and keystone species
Taxonomy of poorly defined taxa in tropical forests: fungi, invertebrates, flora
Functional relationship between animals and ecosystem processes
Gene diversity
Development of monitoring methods and technologies

ANNEX IV

INDICATIVE LIST OF MAJOR SOURCES OF INFORMATION:

Sources of information on the status of forest biological diversity include those provided by governments, inter-governmental organizations (IGOs) and environmental non-governmental organizations (ENGOs) such as:

National Biological Diversity Reports – results from national reporting (CBD article26), and national biological diversity strategies and action plans (NBSAPs; CBD, Article 6) elaborated within the framework of the Convention. The Global Environmental Facility has provided support to 120 countries to assist in the preparation of NBSAPs and their first national report. The 114 national reports received by the Secretariat of the CBD are available at www.biodiv.org. National reports have been received from all regions as follows: Europe (32), Sub-Saharan Africa (28), Asia (14), Middle East/North Africa (9), South America (8), Caribbean (7), Australasia/Oceania (6), Central America (4), Indian Ocean Islands/Madagascar (4) and North America (2). National reports vary considerably in the level of information on biological diversity status that they contain. Most reports include descriptions of the different forest types present, sometimes with information on area and protected status. Information on species diversity is mainly total numbers of species present in broad taxonomic groups, sometimes with information on numbers of endemics and threatened species.

FAO's FRA 2000 report (FAO, 2001a) will incorporate information related to forest biodiversity, including forests in protected areas (see paragraph 102). Information on FRA 2000 is available on line at www.fao.org/forestry/fo/fra/index.jsp

Expert Centre for Taxonomic Identification (an NGO based in Netherlands in collaboration with UNESCO) is developing a World Biological Diversity Database. This is a continuously growing taxonomic database and information system that aims at documenting all presently known species (about 1.7 million). The WBD [or WBDD?] is currently in test phase and contains approximately 56,000 taxa and is expected to expand by an additional 50,000 taxa per year. The database contains taxonomic information (hierarchies), species names, synonyms, descriptions, illustrations and literature references when available.

Global Mangrove Database and Information System (GLOMIS), being developed by the International Society for Mangrove Ecosystems with support from ITTO.

Global Biodiversity Assessment, V.H. Heywood, Executive Editor, R.R. Watson, Chair, United Nations Environmental Programme, Cambridge University Press, 1995.

Functional Role of Biological Diversity, Scientific Committee on Problems on Environment (SCOPE), Mooney *at al.*, 1996.

Various research programmes aimed at examining the effects of management and other interventions on FBD, the evolutionary processes shaping genetic diversity and the relationships between biological diversity, forest functions, socio-economic development and sustainable livelihoods.

Reports from the IUFRO units on Biological Diversity and Population, Ecological and Conservation Genetics.

IUFRO - The Global Forest Information Service (GFIS) arises from the Proposals for Action of the Intergovernmental Panel on Forests and the Intergovernmental Forum on Forest. The mission of the GFIS Task Force is to implement an internet-based forest information service. The system is based on standardizing meta-information catalogues. The service will improve the dissemination and quality of forest-related data and information on forest resources, forest policy, criteria and indicators for sustainable forest management, research activities and other relevant issues. Members of the GFIS Task Force include: European Forest Institute, CIFOR, Forest Research Institute of Ghana, University of Greenwich, ITTO, CATIE, WRI, IUFRO, Chinese Academy of Forestry, FAO Forest Division, Oxford University, CAB International, WCMC, Finnish Forest Research Institute, EMBRAPA Florestas Brazil, US Forest Service, Canadian Forest Service, and Technical University of Vienna.

The website for GFIS is: http://iufro.boku.ac.at/iufro/taskforce/hptfgfis.htm

Other forest information websites include:

http://www.forest-trends.org/keytrends/trendsinforests.htm

http://www.fao.org/forestry

http://www.unep-wcmc.org/forest/homepage.htm

The main sources of information on the status of *forest genetic resources* at global and regional level are:

Reports of FAO's Panel of Forest Gene Experts, the most recent being the 11th session held in September-October 1999 (FAO 2000c).

Country Reports on the State of Forest Tree and Shrub Genetic Resources, available for a number of countries (see FAO 2000c) and regional status and action plans for North America (Rodgers and Ledig 1996), the Boreal Zone (Anon. 1996b) and Europe (Turok *et al.,* 1998) prepared in the lead-up to the Fourth International Technical Conference on Plant Genetic Resources held in Leipzig, Germany, in June 1996 and more recently for dry-zone Sub-Saharan Africa (Sigaud *et al.*, 1998), the Pacific Islands (Pouru 2000), and Southern Africa (Sigaud and Luanga 2000). At a regional level, the main gaps in information on forest genetic resources (diversity levels, processes and threats) are in South and South-east Asia, Central and South America and humid-zone, equatorial Africa. However, in a number of countries research projects are in progress to address these issues, e.g. Brazil' Embrapa Dendrogene Project on genetic conservation within managed forests in Amazonia, Thailand's Royal Forest Department Forest Genetic Resources Conservation and Management (FORGENMAP) Programme and IPGRI and partners research on genetic processes in India (Western Ghats), Costa Rica and Cameroon.

FAO's Global Information System on Forest Genetic Resources or REFORGEN (http://www.fao.org/forestry/foris/reforgen/index.jsp. REFORGEN includes information on forest genetic resources for use in planning and decision-making at the national, regional and international levels. It presently gathers information from 146 countries on more than 1,600 tree species. A more complete data set on the threat status, both at species and population levels, and conservation measures, both *in* and *ex situ*, of included tree species would enhance REFORGEN's utility for documenting and planning forest genetic resources conservation measures.

ANNEX V

CASE STUDY - DEVELOPING MARKETS FOR ENVIRONMENTAL SERVICES TO SUPPORT FOREST CONSERVATION

The Costa Rica Ecomarkets Project

Introduction

The objective of the Costa Rica Ecomarkets Project is to increase the production of environmental services in Costa Rica by supporting the development of markets and private sector providers for services supplied by privately owned forests. It directly supports the implementation of Costa Rica's Forestry Law No. 7575: providing market-based incentives to forest owners in buffer zones and interconnecting biological corridors contiguous to national parks and biological reserves for the provision of environmental services relating to carbon sequestration, biodiversity conservation, scenic beauty, and hydrological services.

Objectives

The project aims to contribute to environmentally sustainable development in Costa Rica through:

(a) supporting the supply of and demand for environmental services provided by forest ecosystems;

(b) strengthening management capacity and assuring financing of public sector forestry programmes administered by the Ministry of Environment and Energy (MINAE), including the National Forestry Financing Fund (FONAFIFO) and the National System of Conservation Areas (SINAC);

(c) increasing inflows of private capital into the forestry sector, sustaining natural forests which are critical for biodiversity conservation and which form the basis for existing (e.g., ecotourism) and emerging industries.

The global environmental objective of the project is to foster biodiversity conservation and preserve important forest ecosystems through conservation easements on privately owned lands outside protected areas in the Mesoamerican Biological Corridor (MBC) in Costa Rica.

Key performance indicators

Key performance indicators related to development include:

(a).............. 30% increase in number of providers of environmental services by end-of-project

(b) 25% increase in land area covered by Environmental Service Payments (ESP) programme contract;

(c)..................................30% increase in the participation of women land owners in the ESP;

(d) .... 30% increase in the participation of women's organizations in the ESP.

Key performance indicators related to the Global Environment objective include:

(a) 50,000 ha of privately owned lands within the MBC incorporated into Costa Rica's ESP programme through conservation easements.

(b) Establishment of a financial instrument to support conservation easements in Costa Rica.

(c) Increased landowner participation in, and benefits from, forest conservation-related activities within the MBC in Costa Rica.

Background

Costa Rica is a leading proponent of environmentally sustainable development, pursuing social and economic growth in conjunction with a strong and healthy environment. The environmental policy of the government has been progressive, including use of economic instruments such as electricity surcharges and reforestation credits which are targeted at protecting forest ecosystems throughout the country. Nonetheless, Costa Rica was beset with one of the highest rates of deforestation worldwide during the 1970s and 1980s. In 1950, forests covered more than one-half of Costa Rica; by 1995, forest cover declined to 25% of the national territory. Approximately 60% of forest cover, totalling 1.2 million ha, exists on privately owned lands outside protected areas. World Bank estimates indicate that 80% of deforested areas, nearly all on privately owned lands, were converted to pastures and agriculture. Deforestation was principally driven by inappropriate government policies including cheap credit for cattle, land-titling laws that rewarded deforestation and rapid expansion of the road system. These policy incentives have since been removed.

Costa Rica's efforts to internalise environmental values provided by forest ecosystems dates back to 1979, with the passage of the first Forestry Law and the establishment of economic incentives for reforestation. Subsequent laws strengthened incentives for reforestation, broadening opportunities for landowners to participate in reforestation programmes and making the programmes accessible to small landowners within rural areas. In 1996, Costa Rica adopted Forestry Law No. 7575, which explicitly recognizes four environmental services provided by forest ecosystems:

(a) mitigation of greenhouse gas emissions, such as CO_2;

(b) hydrological services, including provision of water for human consumption, irrigation and energy production;

(c) biodiversity conservation; and

(d) provision of scenic beauty for recreation and ecotourism.

The law:

(a) delegates responsibilities and duties *inter alia* to licensed forestry regents and municipalities, the National Forestry Financing Fund (FONAFIFO), the National System of Conservation Areas (SINAC) and the Costa Rican Office for Joint Implementation (OCIC);

(b) provides the legal and regulatory basis to contract with landowners for environmental services provided by their lands and establishes a financing mechanism for this purpose;

(c) empowers FONAFIFO to issue such contracts, subject to provisions such as the availability of a forest management plan certified by a licensed forest regent, for the environmental services provided by privately owned forest ecosystems.

With the passage of Forestry Law No. 7575, the forestry sector has an established modern legal framework, which recognizes environmental services provided by forest ecosystems. It defines the role of the State in protecting forests as well as in promoting and facilitating private sector activities, decentralizes duties and responsibilities to local actors, including licensed forestry regents, municipalities and regional councils, and establishes that forests may only be harvested if there exists a forestry management plan that complies with the criteria for sustainable forestry as approved by the State.

Progress so far

The Environmental Service Payments programme, executed through FONAFIFO in close coordination with SINAC, aims to protect primary forest, to allow secondary forest to flourish and to promote forest plantations to meet industrial demands for lumber and paper products. These goals are met through site-specific contracts with individual small- and medium-sized farmers. In all cases, participants must present a sustainable forest management plan certified by a licensed forestry regent, as well as carry out sustainable forest management activities throughout the life of individual contracts. Management plans include information on land cadastre, cartography and physical access; description of topography, soils, climate, drainage, actual land use and carrying capacity with respect to land use; plans for prevention of forest fires, illegal hunting and illegal harvesting; and monitoring schedules. Commitments associated with the environmental service contracts are registered with the deed to the property, such that contractual obligations transfer as a legal easement to subsequent owners for the life of the contract. Furthermore, landowners cede their rights to sequestered carbon to FONAFIFO to sell on the international market.

Environmental service contracts are based upon the value of various services provided by primary and secondary forests, based in part upon studies conducted by the Costa Rica-based Tropical Science Center (see Table 1) and the World Bank (see Table 2). Regulations within Forestry Law No. 7575 establish the conditions for contracting environmental services. Contracts include:

(a) Forest conservation easements: US$220 per hectare disbursed over a five-year period. Eighty-six percent of environmental service contracts in the FONAFIFO programme to date support forest conservation easements, which in large part are targeted at minimizing disturbance of vegetative cover in primary and mature secondary growth forest areas.

(b) Sustainable forest management: US$342 per hectare disbursed over a five-year period. Nine percent of contracts in the FONAFIFO programme support sustainable forest management.

(c) Reforestation: US$560 per hectare disbursed over a five-year period. Landowners must make a commitment to maintain reforested areas for a period of 15-20, depending upon tree species. Five percent of contracts in the FONAFIFO programme support reforestation of degraded and abandoned agricultural lands.

For practical purposes, the ESP programme supports the implementation of Forestry Law No. 7575 by allowing the government to act as a market intermediary: FONAFIFO purchases environmental services from private landowners (e.g., carbon sequestration, biodiversity conservation, hydrological services) and, in turn, sells these services to specific sectors which benefit from these resources.

Table 1. *Minimum, Medium and Maximum Annual Value (1996 US$/ha) for Environmental Services from Primary and Secondary Forests (Tropical Science Center, 1996)*

Environmental Service	**Primary Forest**			**Secondary Forest**		
	Min.	Med.	Max.	Min.	Med.	Max.
Carbon	19	38	57	14.6	29.3	43.9
Hydrologic	2.5	5	7.5	1.3	2.5	3.8
Biodiversity	5	10	15	3.8	7.5	11.2
Ecosystem	2.5	5	7.5	1.3	2.5	3.8
Totals	29	58	87	21	41.8	62.7

Table 2. Estimated Annual Environmental Values (1989 US$/ha) of Primary Forests (Constantino and Kishor, 1993)

Environmental Service	**Primary Forest**	
	Min.	Max.
Carbon Sequestration (about US$20 per tonne of carbon)	60	120
Hydrologic Benefits	17	36
Ecotourism	13	25
Future Pharmaceuticals	0.15	0.15
Funds transfers for existence and option values	13	32
Totals	102	214

Table 3. Total Area and Number of Participants in Environmental Service Payments Programme by Year

Year	**Forest Conservation**	**Sustainable Forest Management**	**Reforestation**

	Has.	**Nº of landowners**	**Has.**	**Nº of landowners**	**Has.**	**Nº of landowners**
1995	23,683	423	--	--	--	
1997[31]	94,484	1,058	8,449	88	4,782	462
1998	46,391	762	8,663	88	4,470	333
Totals	164,558	2,243	17,112	151	9,252	795

[Check number of landowners]

From a conservation perspective, FONAFIFO provides market-based incentives to conserve natural forest ecosystems. These economic incentives help maintain habitats that are critical to a rich, globally important biodiversity and they have the potential to help to maintain biological corridors linking protected areas. Approaching biodiversity conservation through the FONAFIFO mechanism is akin to the system of conservation easements that are widely used in the United States and European countries. In 1997 and 1998, US$15 million were disbursed by FONAFIFO through the ESP programme for the conservation and sustainable use of privately-owned forests; since 1995, over 190,000 hectares of forests have been incorporated into the programme (Table 3) at a cost of approximately US$47 million.

In conclusion, with the introduction of a variety of forest incentives in recent years, Costa Rica has slowed the rapid pace of deforestation witnessed in the 1970s and 1980s. In terms of overall land cover, the gross area of deforestation has been counterbalanced by regrowth in 75% of the previously deforested areas, including the establishment of forest plantations and spontaneous regeneration of abandoned pasture on poor terrain especially in the Pacific slopes. While this regrowth may provide valuable environmental and economic services, it should be noted that, in terms of biodiversity values, it is not equivalent to lost primary forest.

Lessons learned for the next phase of the project

One of the most important lessons learned from activities associated with the projects within the Mesoamerican Biological Corridor is the importance of involving local populations and institutions (e.g. local government, community and sectoral organizations, NGOs) in the design, implementation and benefits of the project in order to assure the long-term conservation of the biodiversity outside protected areas. The project supports the inclusion of small landowners in the ESP programME and technical support for NGOs to provide assistance to small landowners and rural women's organizations relating to forest conservation and sustainable resource management.

A World Bank review of deforestation in Costa Rica carried out in the early 1990s identified three principal types of forest intervention in Costa Rica:

(a)..clear cutting to change the use of lands under forest cover;

(b) ....selective cutting of large, valuable trees in primary or secondary forest;

(c)................................exploitation by owners of pasture areas that contain patches of forest.

The study confirmed that clear-cutting and selective logging are driven by economic interests. While loggers do play an important role, the main motivation for these processes comes from landowners who wish to obtain revenue from the sale of timber or who wish to use the land for agriculture, or both. Environmental concerns tend not to be taken into account by the owners when they are not related to on-site productivity. Hence, the introduction of economic incentives is required where the maintenance of forest ecosystems is considered of importance for the country.

The experience of projects throughout the MBC with buffer zone communities indicates the importance of:

(a) clearly defining the roles of the project and the communities in project administration, fund management, decision-making, and implementation in order to avoid creating false expectations or leaving ambiguities which cause implementation delays;

(b) providing for a strong administrative and coordination capacity supported by adequate technical assistance and, initially, close implementation supervision; and

(c)...................... establishing clear linkages between conservation and development activities.

Phase 2: project summary

Project aims

The next phase of the project will be supported by GEF (World Bank). It aims to increase the production of environmental services in Costa Rica by supporting the development of markets and private sector providers for services supplied by privately owned forests, including protection of biological diversity, greenhouse gas mitigation and provision of hydrological services. As such, the project will support the implementation of environmental policies in the forest sector and contribute to sustainable human development. Additionally, the project will strengthen offices within the Ministry of Environment and Energy (MINAE) as well as local and regional non-governmental organizations (especially women's organizations) responsible for the execution, promotion, supervision and monitoring of the forest conservation programme.

Costa Rica's pioneering efforts to achieve environmental goals through the sustainable use of forest ecosystems entails developing commercially viable activities, which are based upon the environmental services provided from the nation's forests. The project will assist in developing markets, attracting financing and investment and consolidating the institutional framework for:

(a) marketing global environmental services relating to the conservation of biodiversity in privately owned buffer zones surrounding protected areas, thereby protecting the Costa Rican portion of the Mesoamerican Biological Corridor;

(b) marketing global environmental services relating to the mitigation of greenhouse gases, through the development of forestry projects promoting carbon sequestration;

(c) marketing local environmental services provided by forest ecosystems relating to protection of water quality and dry season stream flows in watersheds where hydroelectric projects are presently operating or planned.

Key policy and institutional reforms to be sought

The project will assist in mainstreaming Costa Rica's environmental policies into the forestry sector. With respect to Forestry Law No. 7575, the project will support the contracting of conservation easements for a period of twenty years under Article 22. To date, FONAFIFO has only contracted conservation easements under Article 69, for a period of five years. Under GEF co-financed conservation easements in Tortuguero, La Amistad Caribe and Osa Peninsula, these areas will have a contractual obligation of twenty years. In return for this commitment on the part of small- and medium-sized landowners in these three Conservation Areas, these landowners will receive highest priority for contracts for conservation easements. Furthermore, the Government of Costa Rica is committed to seek continued financing for these conservation easements throughout their twenty-year life.

Benefits and target population

The project should:

(a) empower small- and medium-scale private land owners in the conservation and management of forest ecosystems and in making choices that contribute to sustainable development;

(b) support the long-term viability of the ESP programme and promote increased institutional efficiency of FONAFIFO, SINAC, local NGOs promoting biodiversity conservation, and private sector associations; and

(c) benefit regional users of hydrological services by supporting the provision of high water quality and hydrological stability from forest ecosystems.

Important project benefits include the conservation and sustainable use of forest ecosystems in privately owned land outside protected areas. Replication of programme activities in other countries, including development of markets and private sector providers for environmental services, could further expand project benefits.
