The Precautionary Principle and Forest Exploitation: Implications for the Implementation of the FSC Principle 9.

Janet Cotter H, Paul Johnston, David Santillo
Greenpeace Research Laboratories,
University of Exeter,
EX4 4PS

Greenpeace Research Laboratories Technical Note
No: 08/00, November 3rd 2000

Abstract

Only one half of the original, global forest cover remains, and only one fifth of this forest cover is in the form of what can be regarded as self-sustaining, natural ecosystems (described as frontier forests). In order to preserve the global, regional and local ecosystem service that they provide, the conservation of ancient (including frontier, primary and virgin) forests must be seen as a major theme underlying sustainable development. In order for forests to be sustainably managed, human activities should have no net impact on the ecological integrity of the forest. However, there are major scientific uncertainties regarding loss of biodiversity and ecosystem function. This paper describes the major sources of these uncertainties and points the way towards a strategy incorporating the precautionary principle into forestry management. Any proposed new logging or other large scale operations in ancient forests should be preceded by an independent study which maps and records zones according to ecosystem structure, composition and function. Where permitted, logging and other large scale activities in these regions should move towards the target of sustainable operation, whereby the rate of biomass removal is limited to the rate of regrowth to maturity of the species exploited and the overall structure and diversity of the ecosystem is fully retained.

Introduction

a) Global Deforestation

It has been estimated that of the forest originally present under the prevailing climatic regime and preceding the growth of human influence, around half has disappeared (WRI, 1998). This has been largely due to human activities including increasing arable and livestock agriculture, timber and fuel-wood extraction and other land-use changes consequent to population growth (Groombridge and Jenkins, 2000). Moreover, this loss expressed in absolute terms hides the fact that the remaining forests themselves have been heavily altered by human activity, often reduced to a patchwork of forested areas which differ markedly from continuos forest in composition and ecology (Laurance & Bierregaard, 1997; Noss & Cooperrider, 1994). Hence, only around one fifth of the Earth's original forest remains in the form of large tracts which can be regarded as natural ecosystems, in what have been described as "frontier forests" (WRI, 1998).

The globally aggregated figures hide considerable regional differences, both historical and current. The temperate forests of Europe have been reduced by greater than half, but this reduction largely took place 7000-5000 years ago as agriculture expanded in Neolithic times. Further pressure up to the 11th century reduced forest cover in the UK by 80%. Eastern and Central Europe witnessed a greatly increased rate of deforestation in the 16th and 17th centuries with around 1 million square kilometres being cleared to make way for settled agriculture by 1980 in the former USSR alone. In North America, indigenous populations began to have impacts on forested areas from around 12,000 years ago, but forest cover in the east of this region reached a minimum around 1860 after European settlement. Partial recovery was due to increasing westward expansion resulting in severe impacts through to the early 20th century in forests west of the Appalachians, with pressures continuing due to demand for pulpwood and timber. (Groombridge & Jenkins 2000).

In Oceania, and in tropical Asia, Africa and Latin America, deforestation accelerated markedly as a result of European colonisation. In Oceania, around a quarter of a million km^2 of forest and around half this amount of woodland were lost in the 120 years to 1980. An estimated million km^2 of forest and a similar amount of tropical woodland were converted to crop-growing between 1860 and 1980 in tropical Asia, Africa and Latin America, with the greatest losses in south and south-east Asia (Groombridge and Jenkins 2000).

Currently, rates of deforestation continue to be high in the developing countries of the tropics with the highest rates of a reduction of 1.3% *per annum* (around $10{,}370 km^2$) recorded in Central America, Mexico and the Caribbean. A rate of 0.7% *per annum* reduction (around $36{,}950\ km^2$) is recorded for tropical Africa and of 0.6% *per annum* (around $46{,}550\ km^2$) for tropical South America (Groombridge & Jenkins 2000). The precise methods used to derive these estimates are not always clear, and hence these figures are likely to be significant underestimates in many cases (see: WRI, 1998). Even in those (generally developed) regions where forest cover has increased in recent years, the quality of the regained system (usually plantation) is inferior to natural forest. The condition of forests in these regions is, as in Europe for example, thought to have worsened as a result of fire, drought, pest attack or air pollution (WRI, 1998).

It has been estimated that on a global basis (WRI, 1998) that some 39% of the remaining frontier forest is under moderate to high threat with around three quarters of this being threatened by logging. Mining, roads and infrastructural developments directly threaten almost 40% of the total area facing such compromises with agricultural clearing accounting for another 20%. 27% of the forests under threat face a variety of other negative impacts including excessive vegetation clearance. The fact that these figures add up to a figure greater than 100% illustrates that some areas face ecological damage as a result of multifactorial threats. For example, there is a clear relationship between road development and subsequent forest destruction as a result of the greater accessibility of previously remote areas.

In addition to existing environmental threats, emerging problems are also likely to prove of considerable significance. It has recently been estimated that land-use change followed by climate change are the highest ranking global drivers of biodiversity changes in terrestrial ecosystems and that climate change is likely to be particularly important for boreal ecosystems, including forests (Sala *et al.*, 2000). Recent modelling predictions (DETR 1998) suggest that by the 2050s, many regions which currently support tropical forests will change to savannah, grassland or even desert. The ecosystem modelling predicts that this dieback will occur over vast areas of northern Brazil, beginning in the 2040s, resulting from decreases in rainfall of up to 500 mm per year and increases in temperature of up to 7 °C.

Gauged against the comprehensive threats to frontier forest ecosystems, the degree to which they are protected is essentially minimal. Currently, around 8% of global forests are included in protected areas as defined by IUCN management categories I-VI, the majority in category VI. Less than 4% are managed under categories I & II which provide, theoretically, the most rigorous protection (Groombridge and Jenkins, 2000). The actual protection afforded by these schemes, however, is widely regarded as ineffective.

b) Ecological Services

The direct economic values of forest products (see, e.g. Cotton, 1999) are counterpointed by the considerable value of natural capital that they also provide. Forests provide resources in addition to timber including, for example, rubber, foodstuffs, medicinal plants and fibre sources which currently tend to be considerably undervalued relative to extractable timber resources. In short, the support of human existence by forests goes far beyond the simple commercial exploitation of their resources or their more direct support of indigenous populations. Forests are a vital component of the Earth's biosphere. They are major reservoirs of biological diversity, are important in maintaining the carbon balance of the biosphere, in regulating global energy changes and transfers, provide watershed protection and help to regulate nutrient and hydrological cycles. Some attempts have been made to value the ecosystems services provided by each of the earth's biomes. (Costanza *et al.*, 1997). The published figures suggest that, per hectare on an annual basis, forests provide ecological services of some US$969. Tropical forests provide an estimated US$2,007 $ha^{-1}y^{-1}$ while temperate forests contribute US$302 $ha^{-1}y^{-1}$. In aggregate, forests account for approximately US$4.7 billion of services. This represents 14% of the ecosystem services provided by global biomes (Table 1) and around 40% of the ecosystem services provided by terrestrial environments alone. In most cases these services accrue directly to humans without passing through formal monetary economies and any assigned value has therefore to be treated as, at best, a very approximate estimate.

TABLE 1 The global value of ecosystem services calculated for the year 1994. Missing values denote insufficient information to make a calculation. Values were derived on the basis of a limited set of ecosystem services by Costanza *et al.*, 1997.

BIOME	**Area** (ha x 10^6)	**Value** ($US ha^{-1} y^{-1})	**Total Global Value** ($US x 10^9 y^{-1})
Marine	**36,302**	577	**20,949**
Open Ocean	**33,200**	252	**8,381**
Coastal	**3,102**	4,052	**12,568**
Estuaries	180	22,832	4,110
Seagrass/algae	200	19,004	3,801
Coral Reefs	62	6,075	375
Shelf	2,660	1,610	4,283
Terrestrial	**15,323**	804	**12,319**
Wetlands	**330**	14,785	**4,879**
Tidal Marsh/Mangroves	165	9,990	1,648
Swamps/Floodplains	165	19,580	3,231
Forest	**4,855**	969	**4,706**
Tropical	1,990	2,007	3,813
Temperate/Boreal	2,955	302	894
Grass/Rangelands	**3,898**	232	**906**
Lakes/Rivers	**200**	8,498	**1,700**
Desert	**1,925**		
Tundra	**743**		
Ice/Rock	**1,640**		
Cropland	**1,400**	92	**128**
Urban	**332**		
TOTAL	**51,625**		**33,268**

These values are indicative of the considerable reliance of humans upon largely intangible rather than commercial benefits derived from forests. However, they must be treated with a degree of caution. In many cases they derive from "willingness to pay" on the part of individuals for the services or functions in question. A truly accurate valuation by these methods, therefore, is dependent upon such individuals living in a sustainable manner and recognising the extent of their connection to and dependence upon ecological services provided by forests. Moreover, lack of information about many biological systems, distortion introduced into notional values by a variety of factors, assumptions made about supply and demand, differences in national income levels and interdependencies in the ecosystems themselves, all conspire to introduce inaccuracies (Costanza *et al.,* 1997). Such problems encountered with applying contingent valuation techniques to tropical rainforests have been documented by Carson (1998).

There are two further important critical limitations to the use of ecological economic methodologies (Costanza *et al.,* 1997). First, these analyses generally assume that in responding to perturbation, ecosystems will exhibit no sharp thresholds, discontinuities or irreversibilities. This is acknowledged by the authors as a false premise. Second, and more serious, is the misconception that should these ecosystems be utilised in a non-sustainable manner then the assigned value represents the cost of substituting them by technological means. Allied to this is the misconception that environmental damage can be paid for and that this is as good as, or even preferable, to avoiding the damage in the first place (Beder, 1996). These views fail to recognise that without ecological services such as those provided by the forests, sustainable use of the planet would simply not be possible. Some of these functions are, in actuality, irreplaceable (Cairns & Dickson, 1995). These efforts, however, involve pricing and in turn these prices should always tell the truth about the values of the ecological services upon which humanity depends. (Tickell, 1997; Arrow *et al.,* 1995; Costanza *et al.,* 1997). The actual costs of the loss of an irreplaceable ecosystem function could quickly become infinite.

Sustainability and The Precautionary Approach

a) Sustainability

Extending economic analogy leads to the basic truth that current world development is proceeding on the basis of utilising ecological "capital" rather than the "interest" accruing from sustainable utilisation of ecological services (Cairns, 1996). The impacts of many forestry activities cannot ultimately be justified, even through the most severely reduced interpretation of ecological economics, namely the simple precedence of benefit over cost. Put another way, the ecological services provided by global forests as an exemplary system are being overused at current assigned prices (Costanza *et al.,* 1997). Such activities are not sustainable. In this context, the term sustainable refers to the forest ecosystem considered as a whole. Sustainability defined purely in terms of resource extraction from forests has long been a traditional source of tension between forestry managers, with whom maximum yield remains a dominant management paradigm, and ecologists. Ecological values are more grounded in the non-commodity values of forests (see Noss & Cooperrider, 1994).

The need to consider ecosystems in detail with respect to either commodity or non-commodity values leads to the basic paradox that a sustainable system can generally only be identified as such after the fact of exploitation. Accordingly, definitions of sustainability are usually only predictions of the sets of conditions that will actually lead to sustainable systems (Cairns, 1996), rather than robust definitive criteria. Resolving the conundrum of ensuring that ecosystem services are provided at a sustainable rate which meets societal demand without compromising the service for future generations is undoubtedly extremely difficult (Cairns, 1997). As provisional overarching principles of sustainability, however, the following set of four conditions as listed by Cairns (1997) have some merit:

1) *Substances from the earth's crust must not systematically increase in the ecosphere*

2) *Substances produced by society must not systematically increase in the ecosphere*
3) *The physical basis for productivity and diversity of nature must not be systematically diminished*
4) *Fair and efficient use of resources with respect to meeting human needs*

The use of these conditions as predictive or definitional tools is inevitably limited by incomplete or uncertain data relating to any proposed human activity. Nonetheless, they are useful because all are requirements for sustainability and taken together are sufficient to ensure sustainability. Taken as a set of encompassing conditions for sustainability, they can be used conveniently as a checklist against which human activities can at least be retrospectively evaluated in a relatively simple way. Uses of ecosystem services on a sustainable basis should not violate any of these principles. It follows that environmental protection should set standards such that there is a very high degree of certainty that these principles will not be compromised.

The four provisional conditions must be viewed against a background of a continuing increase in the pressures upon forest environments and the ecological services which they provide. It is estimated that already some 75% of the total habitable area of the planet has been disturbed by human activity (Hannah *et al.,* 1994) and world population is set to continue to increase.

Given the certainty of continuing human pressures, it is clear that the dominant paradigms of forest management will need to change. As a starting point, an appreciation of the considerable uncertainties which exist in human understanding of the varied processes normally taking place in forest ecosystems must be developed in order to underpin this change. The uncertainties attached to scientific ability to predict the responses of ecosystems to human interference also need to be strategically recognised. Given this, it follows that concrete strategic outcomes which significantly contribute to achieving greater sustainability and viability of forest ecosystems will need to be formulated on a basis inclusive of the inherent uncertainties. Only then will policies be based upon realistic scientific evaluation and enjoy widespread public confidence.

b) The Precautionary Approach and Risk

The most widely accepted means of protecting the environment in the face of the multifarious uncertainties is through the adoption of a Precautionary Approach (Stirling, 1999). A great number of international fora have adopted a precautionary approach to environmental protection. It is affirmed as a general protective axiom in Principle 15 and Agenda 21 of the 1992 Rio Declaration on Environmental Development. Other agreements espousing a precautionary approach include the Convention on Biological Diversity, the UN Agreement on the Conservation and Management of Straddling and Highly Migratory Fish Stocks (1995), the OSPAR Convention (1992) the 1996 Protocol to the London Convention and the Barcelona Convention as amended in 1995. The approach evolved originally from efforts to regulate and control hazardous chemicals entering the sea (Stairs & Johnston, 1991; Johnston & Simmonds, 1991; Jackson & Taylor, 1992) as exemplified by the 1987 Third Ministerial Declaration on the North Sea (MINDEC, 1987).

The use of a Precautionary Approach resulted from increasing recognition that ecological systems cannot be comprehensively observed and that impacts cannot, therefore, be fully regulated and controlled. Broadly speaking, a precautionary approach recognises scientific and technical limitations and promotes regulatory action in the absence of full evidence of a cause-effect relationship. In short, it allows incomplete data, uncertainty and indeterminacy to be taken into account in a meaningful way in the decision making process (Stirling, 1999). Increasingly, the limitations of conventional impact testing and predictive regimes and of biological surveys are being recognised (see, e.g. Cairns, 1989). Uncertainties in understanding ecosystems are conveniently illustrated in terms of Figure 1. This figure was originally designed to illustrate issues in aquatic ecotoxicology (Santillo *et al*., 1998) but as adapted it is nonetheless highly relevant to forest management. It illustrates that as levels of biological organisation increase from the level of individual organisms to ecosystems, the knowledge concerning important functions declines. For example, the impact upon an individual tree removed by logging or other activities is fairly clear. The implications of the removal of a large number of trees for whole populations and ecosystems, by contrast, cannot be easily determined. Paradoxically, while changes at the organismal level are the current cornerstone of ecological assessments, the protection of populations, communities and ecosystems is of greatest concern to environmental managers. It follows that the choice of both test and assessment endpoints for any given management goal is of critical importance (Suter, 1994). This, of course, assumes that endpoints actually exist in the form of conveniently measurable parameters, and this is not often the case.

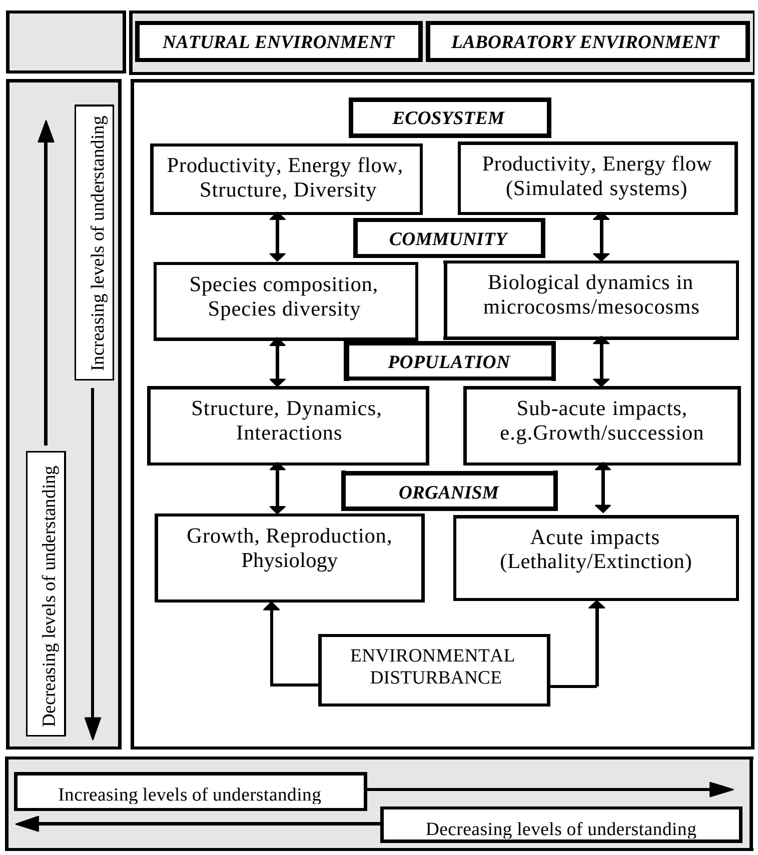

FIGURE 1 Schematic showing relative levels of understanding possible/likely at different levels of biological and ecological organisation. Adapted from Santillo *et al.* (1999).

Overall, the scientific and technical limitations can act to completely undermine attempts at assessing the environmental consequences associated with any given human activity. Far from being an analysis in numerical terms, therefore, environmental risk/impact assessment procedures are increasingly recognised as being based on several serious misconceptions. These misconceptions are best characterised for toxicological assessments (see Power & McCarty, 1997), perhaps explaining the early application of precautionary approaches towards chemical regulation.

Hence, the term "risk assessment" when applied to ecological systems should not be confused with actuarial risk analysis based upon hard data as practiced in the fields of engineering or insurance underwriting. The interpretation of the relevance of, for example, forest ecosystem or forest organism population data requires an insight into the functioning of the whole ecosystem in terms of multiple anthropogenic and natural stressors and the interactions between those stressors (Santillo & Johnston, 1999 a &b). Unfortunately, forest ecologists, in common with those practising in other branches of ecology do not currently know which factors are the most critical (Power & McCarty, 1997). Ecological risk assessment, therefore, can be best viewed, perhaps, as a developing arm of the "prediction industry" which is more traditionally associated with financial markets, meteorological forecasts and personal horoscopes. This industry, with considerable justification, is coming under increased critical scrutiny in relation to the accuracy of the results that it achieves (Sherden, 1998). Considerable problems arise in attempting to define the primary aim of science in terms of prediction. The predictive prospects of science are limited by the role of chance and chaos which create uncertainties (Reschler, 1998). Science undoubtedly has a role in description, classification, evaluation and control of problems. Unless uncertainties are explicitly recognised, a major constraint on scientific prediction is not incorporated into the predictive equation. The necessary alternative to the current permissive regimes of environmental regulation and control in relation to forest exploitation is found in a precautionary approach.

c) The Precautionary Approach Defined

A precautionary approach to environmental protection can be defined as:

The emplacement of appropriate preventative measures when there is reason to believe that harm is likely to be caused by anthropogenic activities including the introduction of substances or energy into the environment and the extraction of species (including non-target species). Action should be taken even where there is not conclusive evidence to prove a causal relationship between the actions and their effects.

Recently, it has been noted (Sandin, 1999) that most formulations of the Precautionary Approach can be distilled into the following simple form:

If *there is (1) a threat, which is (2) uncertain* **then** *(3) some kind of action (4) is mandatory*

In this formulation the prospect of potential harm as outlined in the first definition is also encapsulated in this condensed version. Indeed, the effectiveness of management based upon precaution depends in large measure upon anticipation of impacts. This approach to environmental protection contrasts with measures taken only after harm has been identified (as is generally the case) and allows consideration of all the information available. It effectively reverses the burden of proof and places it upon those seeking to exploit forest resources at the potential expense of ecological services. Unresolved uncertainties compromise both ecological economics and environmental risk assessment as noted above. A precautionary approach allows these areas of uncertainty to be identified and considered explicitly in regulatory processes.

Such an approach, however, has a wider application than the formulation of regulatory instruments for human activities and can be used more holistically. It can be applied to examination of the activities in relation to Cairns' (1997) four provisional conditions of sustainability outlined above. If it cannot be proven that a given activity is not going to violate one or more of these principles, then this activity should be prohibited or at least more tightly controlled. In this way, a precautionary approach to environmental protection can be regarded as an instrument of both regulatory activity and sustainability.

Despite the noteworthy developments in accepting a precautionary approach in various international fora, it has not been universally applied as yet and in the forest sector has been effectively and conspicuously lacking as a management paradigm. Effective implementation of the approach needs to be actively supported, promoted by national Governments. By promoting effective application of such an approach important linkages are more likely to be crystallised among the diverse, but interconnected initiatives which are intended to promote the sustainability and the viability of the forest realm.

Recent initiatives for the management and conservation of forests have been put forward. The Forestry Stewardship Council (FSC) have introduced Principle 9 to their criteria (FSC, 2000) which refers to the "Maintenance of High Conservation Value Forests" (HCVF). Further, the Principle sates "Decisions regarding high conservation value forests shall always be considered in the context of a precautionary approach". Recently (summer 2000), the FSC have requested comments on the "Report of the Principle 9 Advisory Panel: Draft Recommendation". A major part of the Draft Recommendation and associated Background Paper concern the nature and implementation of the precautionary principle.

Forest Conservation

Applying the precautionary principle to forest conservation in relation to the High Conservation Value Forests (HCVFs) specified by the FSC implies that these forests need to be either already identified or that projected developments are evaluated for High Conservation Value properties in order to identify them reliably in advance of proceeding with exploitation. Classification of forests has been approached in a number of ways, and a useful framework is that detailed by Noss & Cooperrider (1994). These workers convincingly argue that primary natural forest has the highest conservation value because it is rare and depleted in most regions. With respect to the regions in which it occurs it is generally regarded as having the maximum biological, ecological and structural diversity. They also regard some secondary natural forest as having a moderate to high conservation value, depending upon its location, while the conservation value of plantation is generally low. This implies that forest landscapes capable of "self management" should be tightly controlled with respect to human uses. In cases of fragmented forest habitat, conservation implies careful management regimes designed to simulate natural phenomena and aimed at the enlargement and connection of forest fragments to increase resilience. It follows from this that all areas of primary natural forest, and specifically of frontier forest, can be regarded as having a High Conservation Value and that management should be strictly aimed at avoiding any breach of the principles of sustainability and conducted, therefore, on a precautionary basis.

In any given situation involving human interaction with frontier forest (intrinsically with a high conservation value) the potential impacts should be thoroughly and exhaustively evaluated. As part of this process, the dimensions of uncertainty and ignorance (including an appreciation of "what we don't know that we don't know") should be explicitly defined. A raft of assumptions is characteristically applied in such evaluative procedures. These, as aspects of uncertainty/ignorance should also be specified in detail. In short, the potential impacts need to be thoroughly explored prior to exploitation and the impact during the exploitation phase should be strictly and comprehensively monitored.

These considerations are implicitly, though less directly acknowledged in the definition of HCVFs by the FSC (1999) which defines them as forest areas with one or more of the following attributes:

a) *forest areas containing globally, regionally or nationally significant concentrations of biodiversity values (e.g. endemism, endangered species, refugia); and/or large landscape level forests, contained within, or containing the management unit, where viable populations of most if not all naturally occurring species exist in natural patterns of distribution and abundance*

b) *forest areas that are in or contain rare, threatened or endangered ecosystems*

c) *forest areas that provide basic services of nature in critical situations (e.g. watershed protection, erosion control)*

d) *forest areas fundamental to meeting basic needs of local communities (e.g. subsistence, health) and/or critical to local communities' traditional cultural identity (areas of cultural, ecological, economic or religious significance identified in cooperation with such local communities).*

Although these properties are intuitively highly attractive as a means of classifying foci of potential exploitation, attribution of one or more of these properties to a given forest ecosystem implies the need for a full characterisation of the system in question. Application of the precautionary principle should be on the basis that, as a governing axiom, all ancient forest should be considered as having globally significant conservation value.

Uncertainty, Ignorance and Forest Systems

The initial identification of a forest ecosystem as HCVF requires a full characterisation of the ecosystem concerned in order to establish the relevance of the FSC designated properties. Explicit identification of uncertainty and ignorance relative to forest systems is a fundamental prerequisite of a precautionary management regime. As noted earlier, natural forests are providers of vital ecosystem services. Hence, the degradation of forest ecosystems results in a deterioration of these services, initially on a local level, but ultimately expressed as a regional and even global concern. A comprehensive failure of these essentially irreplaceable systems could impact, *inter alia* ecological systems, upon the regulation of atmospheric O_2 and CO_2, upon climatic regulation, protection of soil from erosion and upon hydrological and elemental cycles. This leads to the first key area of uncertainty: To what degree can forests be exploited before these ecosystems and their associated services fail?

The functioning of a forest ecosystem can be characterised broadly in terms of its biomass, its biodiversity, and the presence or absence of multiple stressors. Of these three factors, biodiversity is the hardest to quantify, both in terms of quantitative measurement and also in the minimum amount necessary to sustain ecosystem function. The precise characterisation of the stressors acting upon the ecosystem is also typically difficult to achieve. Taken together, these are the major sources of scientific uncertainty and ignorance relative to forest ecosystems. It follows that if these uncertainties and lack of knowledge cannot be resolved then exploitation should not proceed.

a) Biodiversity and ecosystem function

Forests are an important reservoir of biodiversity, arguably the most important reservoir. Ancient and frontier forests, because of their long standing and relatively lower levels of human disturbance, are typically richer in biodiversity than other natural or semi-natural forests. An illustration of the conservation importance of forests relative to biodiversity is found in the recent analysis of biodiversity hotspots (Myers *et al.*, 2000). 25 hotspots were identified as conservation priorities on the basis of species endemism and the degree of threat through habitat loss. Although the "hotspot" approach has many limitations in the identification of conservation priorities, a noteworthy feature of this analysis was the fact that tropical forests featured predominantly in the list, comprising 15 hotspots with Brazil's Atlantic Forest featuring in the top five.

Loss of forest will inevitably result in reduction of biodiversity as a direct result of loss of habitat. There are many anthropocentric reasons to preserve biodiversity (direct use value, option value, etc.) but the principal such reason considered here is the indirect use value in the form of ecosystem services. A loss in biodiversity affects the stability of an ecosystem resulting in a reduction of its resistance to disruption of the food web (by loss of the weak interaction effect), resistance to species invasion and resilience to global environmental change (McCann, 2000; Chapin *et al.*, 2000).

The argument for conserving forest biodiversity is, therefore two-fold:

1) biodiversity is essential for maintaining ecosystem services and

2) biodiversity increases the stability and resilience of an ecosystem to a disturbance (e.g. climate change).

Given that the ecosystem services of forests are critically dependent on biodiversity the central question then becomes: "For an individual tract of forest, how much loss in biodiversity can be withstood before ecosystem function is severely impaired?" This is likely to prove difficult to answer in definitive terms but several theoretical models relating species richness to ecosystem function have been proposed (Gaston & Spicer, 1998). Virtually all models show a decline in ecosystem function as species numbers decline. Those that don't respond in this way indicate that change is highly unpredictable. The principal difference between models is whether the decline is linear or not. Non-linear models show little or no decline up to a threshold point followed by a sharp decline or catastrophic failure of ecosystem function. The question then arises as to what is the threshold point? The threshold point could be the loss of a keystone species or loss of a key functional grouping. But, would the identity of the keystone species be known in advance? Reports of ecosystem perturbation in the literature caused by the loss of a keystone species have only been only made with hindsight, e.g. sea otters (Estes & Palmisano, 1978; see also Santillo *et al.,* 1998). Which species are the most important to protect in terms of ecosystem function simply cannot be decided on the basis of current knowledge. Indeed as a basic illustration of this, the IUCN Red List of Threatened Species (IUCN, 2000) lists species in terms of their abundance, not in terms of their importance to ecosystem function.

It would be rather fruitless to try and identify forest keystone species and expect to do so with any reliability. This simply follows from a general lack of knowledge regarding forest ecosystems. Scientific knowledge is limited regarding the functional nature of individual inter-species relationships. Relatively few studies have reported definitive associations between species such as the pollination of individual species of the fig by a single species of wasp (Janzen, 1979). It is generally accepted, however, that there are a multitude of inter-species relationships operating within and between all trophic levels in forests (e.g. the association of trees and mycorrhizas) because of the long history of co-evolution (c. 400 Ma for plants and insects). It therefore follows that there are many inter-species relationships that have not yet been identified or qualified, much less characterised and quantified. Moreover, the are undoubtedly many species that have not yet been "formally" discovered, e.g. it is estimated that only 5 % of fungi and 0.4 % of bacteria have been identified (Hawksworth & Kalin-Arroyo, 1995). Hence, in terms of defining keystone species of ecosystem function for forests, as with many other ecological systems, we can only profess ignorance and uncertainty.

These, admittedly limited, examples illustrate that for forest ecosystems the various fundamental uncertainties relating to whole ecosystem assessment methodologies in general as outlined earlier in this document are fully relevant. As well as the ecological value of biodiversity, it is becoming increasingly accepted that quite apart from commercially valuable species diversity of life has an intrinsic moral as well as monetary value (O'Niell, 1997; Oksanen 1997; Moyle & Moyle, 1995). An important conclusion that arises from this is that if such ecological aspects and attributes cannot be reliably characterised in terms of likely departures from baseline conditions, development and exploitation should not be sanctioned. Put simply, there is no answer to the central question "How much biodiversity is enough?" considered in terms of whole ecosystem function and ecosystem services. This serves to emphasise further the extreme global need to conserve frontier forest.

b) Biodiversity, Fragmentation and Extinction

As well as being a key determinant of ecosystem function, biodiversity affects the ability of an ecosystem to recover from disturbances and to respond to environmental perturbation (Chapin *et al.,* 2000). Biodiversity is effectively a natural insurance policy against such environmental changes. Habitat loss (through deforestation) has a direct effect on the ability of species to respond to environmental change on local, regional or global scales. The ability of a species to survive climate change, for example, can be related to the rate at which it can spread to new habitats. If a forest habitat is fragmented or lost through land-use change, the area that it can move within and to is restricted. This is particularly relevant to frontier forests whose large tracts of undisturbed forest allow the migration of species (e.g. the boreal forest of Russia).

Fragmentation of these areas will result in a potential loss in biodiversity (see: Noss & Cooperrider 1994).

Fragmentation also leads to genetic isolation of plants and animal species, reducing genetic biodiversity of species. This especially true of species with a wide range, e.g. large mammals (Foreman and Collinge, 1996), which are vulnerable to fragmentation, not only from genetic isolation but also as it restricts their ability to roam in search of food. Ultimately, this could result in a loss of large mammal species, reducing biodiversity. Fragmentation also effects the trees themselves as. For example, Laurance *et al.* (2000) found that not only did more trees die near forest edges but that a higher proportion of dying trees were large. Large, mature trees are important for animal shelter and reproduction. Seed germination in rainforest fragments has been shown to be impaired, with seeds in fragments suffering from edge effects including hotter, drier conditions and increased light penetration (Bruna, 1999).

Ultimately, restriction of a habitat can lead eventually to local extirpation of a species and its eventual extinction. Current rates of extinction of species vary from between 100-1,000 times (Jablonski, 1995) to 120,000 times (Myers, 1993) greater than the pre-human rate. Although there is a huge variability in these estimates, extinction rates are obviously much higher now than the pre-human era. Species extinctions are irreversible. There are huge uncertainties also as to the effects of such high extinction rates: on a commercial basis, species that are potentially useful to humans (medicinal or economic) may be eliminated. In ecological terms, keystone species may be eliminated causing a cascade of linked extinctions, altering the food web (Myers, 1993). Extinction itself obviously leads directly to a loss in biodiversity (as opposed to loss of diversity through loss of habitat) with the poorly understood but negative implications for ecosystem function described above. From an examination of the fossil record through the Phanerozoic, Kirchner and Well, (2000) and Jablonski (1995) estimated that rates of origination of new species took approximately 10 Ma to recover following extinctions (both major and background). If we estimate that *Homo sp*. have existed for 2 Ma, this recovery time is well beyond the human time scale. Accordingly, any vital ecosystem function which fails as a result of keystone extinction(s) thereby fatally compromising the integrity of ecosystem services (i.e. giving them an infinite value as discussed above) is unlikely to re-evolve within human time scales.

Again, these limited examples, together with the extensive literature detailing the impacts of fragmentation on biodiversity (see: Noss & Cooperrider, 1994; Laurance and Bierregaard, 1997) serves to underline the importance of conserving frontier forest as a matter of high priority.

Sustainability and Precaution applied to Forest Systems

From the foregoing discussion it can be concluded that the identification and conservation of frontier forests is an important imperative and that all frontier forests have the attributes of HCVFs. At the same time, it is clear that scientific ability to characterise such systems in ecological terms is fraught with a very large number of potential uncertainties and indeterminacies.

These uncertainties may be distilled down into two key areas. Firstly, it is not known how much biodiversity is necessary to maintain forest ecosystem services or allow systems to adapt to environmental change and which are the keystone species deserving conservation priority.

Secondly, we do not understand with any certainty the impact of forest exploitation upon biodiversity, although forest fragments are regarded as generally less biodiverse than primary forest.

Accordingly, management strategies need to be set on a precautionary basis such that loss of biodiversity is eliminated in frontier forest areas. This may be seen as the only reliable way of sustaining forest ecosystems at their current functional level in order to be certain to avoid catastrophic collapse of forest ecosystem services. Stewardship of frontier forest ecosystems needs to shift rapidly away from exploitation of the natural capital towards generalised protection associated with sustainable (limited) use. "Use" must be recognised as extending well beyond the simple extraction of wood. Ultimately, the goal should be the maintenance of the current standing biomass and level of biodiversity.

A truly precautionary management approach, therefore, should aim to rigorously satisfy the four principles of sustainability outlined above. Such an approach must take into account *inter alia* scientific information with respect to the potential for harm, as well as the uncertainty, indeterminacy and ignorance integral to our knowledge of the structure and function of natural systems. Implicit also to a precautionary approach is the goal to reduce progressively overall human impacts on the biosphere.

Elements of a Precautionary Approach to Sustainable Forests

Of the first order principles for sustainability outlined by Cairns (1997) numbers 3 and 4 are particularly relevant to the exploitation of forest ecosystems. As reproduced below, they provide an effective benchmark against which the sustainability of any particular practice or strategy may be gauged.

3. *The physical basis for productivity must not be systematically diminished.*
4. *Resources must be used fairly and efficiently with respect to human needs.*

With respect to ancient forests, these principles can be seen to be reflected in the following more specific measures which are highly precautionary in their intent.

To ensure that the physical basis of productivity (and therefore biodiversity) is not systematically depleted:

1. No new logging or other large scale operations in ancient forests should be permitted until such time as an independent study of ecosystem structure, composition and function is completed. This should include species abundance/biomass and diversity at all trophic levels, plus consideration of abiotic factors such as soil structure and hydrological balance. The study would necessarily involve comparison with an area of forest with comparable natural history as a control.

2. If this study indicates that biodiversity (including the abundance of rare and/or sensitive species) at the proposed or active site is threatened by the activity, or has already been depleted as a result of ongoing logging or other operations, no new or further logging or other large scale activities should be permitted.

3. Data acquired through survey and study programmes conducted directly or by using remote sensing techniques should be used to construct comprehensive maps of ancient forest areas and to zone such areas according to their determined conservation value. If development and exploitation of such resources is permitted in a zone subsequent to the assessment process, then this should be conducted in accordance with points 4-9 below.

4. Any new or existing permitted operations should be subject to periodic review, including re-evaluation of status and threats to biodiversity, such that the potential for any emerging systematic degradation can be avoided.

5. Clear-felling should not be permitted as part of any logging operation in ancient forests – individual trees should be selectively felled such that a proportion of the initial biomass remains (including also the understory) sufficient to preserve ecosystem structure, function and biodiversity.

6. The removal of dead trees should be limited to allow a proportion to remain sufficient for the maintenance of invertebrate populations.

7. In any operation, a set proportion, and absolute biomass, of the current forested area should be subject to preservation, with no logging or other large scale activities permitted. Protected areas should be clearly defined in terms of boundary co-ordinates and their preservation subject to international oversight.

8. No logging or other activity should result in the creation of forest islands, whereby protected areas are surrounded by areas subject to exploitation. Protected areas in any one forested region must be defined in order to ensure contiguity and permit unimpeded passage for wildlife within protected areas.

9. Regular reporting requirements should include precise co-ordinate definition of areas subject to logging or other exploitation. Records should be maintained of disturbances including species and biomasses removed. Logging and transport methods employed should be subject to international verification.

To ensure fair and efficient use of resources with respect to meeting human need:

1. All new or existing operations must permit continued legitimate and sustainable human usage of forest areas by indigenous peoples, without requirements for displacement and relocation. All proposed or ongoing operations with the potential, realised or otherwise, to impact on indigenous and other traditional communities should be subject to detailed consultation with those communities, as well as with local authorities, including the provision of access to independent legal advice where appropriate.

2. Regular reporting requirements should include specific tonnages processed and used in the state of origin, specific tonnages exported, countries of import and all end use categories.

Conclusion

All remaining ancient forests should be regarded as of high conservation value in recognition of the vital role they play in local, regional and global biosphere processes and in maintaining biodiversity. The conservation of ancient (including frontier, primary and virgin) forests must be seen as a major theme underlying sustainable development. Accordingly, logging and other large scale activities in these regions should move towards the target of sustainable operation, whereby the rate of biomass removal is limited to the rate of regrowth to maturity of the species exploited and the overall structure and diversity of the ecosystem is fully retained. In the meantime, ancient forests should not be subject to industrial exploitation until the conservation value has been properly assessed and the forests themselves zoned according to the properties established on this basis. In turn, based upon this, an effective system of protected areas needs to be successfully established. In order for a forest to be sustainably managed, human activities should have no net impact on the ecological integrity of the forest. Since inter-species functional relationships and those between species and their environment are poorly known, the precautionary principle must be employed to ensure that ecological integrity is maintained in these forests.

References

Arrow, K., Bolin, B., Costanza, R., Dasgupta, P., Folke, C., Holling, C.S., Jansson, B.O., Levin, S., Maler, K.-G., Perrings, C. & Pimentel, D. (1995) Economic growth, carrying capacity, and the environment. *Science,* **268**: 520-521.

Beder, S. (1996) Charging the earth: the promotion of price-based measures for pollution control. *Ecological Economics,* **16**: 51-63.

Bruna, E.M. (1999) Seed germination in rainforest fragments. *Nature*, **402**: 139.

Cairns, J. (1989) Applied ecotoxicology and methodology. In: Boudou, A. & Ribeyre, F. (eds.) *Aquatic Ecotoxicology: Fundamental Concepts and Methodologies* CRC Press, Inc. Boca Raton pp275-290.

Cairns, J. (1996) Determining the balance between technological and ecosystem services. In: Schulze, P.C. (ed.) *Engineering within Ecological Constraints.* National Academy Press, Washington, pp13-26.

Cairns, J. (1997) Defining goals and conditions for a sustainable world. *Environmental Health Perspectives,* **105** (11): 1164-1170.

Cairns, J. & Dickson, K.L. (1995) Individual rights versus the rights of future generations: ecological resource distribution over large temporal and spatial scales. Chapter 11. In: Ingman, S.R., Pei, X., Ekstrom, C.D., Friedsam, H.J. & Bartlett, K.R. (eds.) *An Ageing Population, an Ageing Planet, and a Sustainable Future.* Texas Institute for Research and Education on Ageing, Denton, Texas.

Carson, R.T. (1998) Valuation of tropical rainforests: philosophical and practical issues in the use of contingent valuation. *Ecological Economics,* **24**: 15-29.

Chapin III, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C. & Diaz, S. (2000) Consequences of changing biodiversity. *Nature,* **405**: 234-241.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O,Niell, R.V., Paruelo, J., Raskin, R.G., Sutton, P. & van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature*, **387**: 253-260.

Cotton, C. (1999) Buying Destruction: a Greenpeace Report for Corporate Consumers of Forest Products. Greenpeace International, Amsterdam, ISBN 90 73361-54-0 pp76.

DETR (1998) Climate Change and its Impacts: Some Highlights from the Ongoing UK Research Programme: a First Look at Results from the Hadley Centre's New Climate Model. The Meteorological Office/United Kingdom Department of Environment, Transport and the Regions.12pp

Estes, J.A. and Palmisano, J.F. (1974) Sea otters: their role in structuring nearshore communities. *Science*, **185:** 1058-1060.

FSC Forestry Stewardship Council (2000) Principles and Criteria v. 1.2 www.fscoax.org

Foreman and Collinge, S.K. (1996) The "spatial solution" to conserving biodiversity in landscapes and regions. In: DeGraaf, R.M. and Miller, R.I. (eds.) *Conservation of Faunal Diversity in Forested Landscape*. Chapman and Hall, London. pp. 537-568.

Gaston K.J. and Spicer, J.L. (1998) *Biodiversity: an Introduction.* Blackwell Science, Oxford.

Groombridge, B. & Jenkins, M.D. (2000) Global biodiversity: Earth's living resources in the 21st century. United Nations Environment Program- World Conservation Monitoring Centre Publ World Conservation Press, Cambridge, 246pp

Hannah, L., Lohse, D., Hutchinson, C., Carr, J.L. & Lakerani, A. (1994) A preliminary inventory of human disturbance of world ecosystems. *Ambio* **23**: 246-240

Hawksworth, D.L. and Kalin-Arroyo, M.T. (1995) In: Heywood, V.H *(ed.) Global Biodiversity Assessment* Cambridge University Press, Cambridge, pp.107-191.

IUCN (2000) 2000 IUCN *Red List of Threatened Species.* IUCN, Gland Switzerland.

Jablonski, D. (1995) Extinctions in the fossil record. In: May, R.M. and Lawton, J.H. (eds.) *Extinction Rates.* Oxford University Press, Oxford. pp. 25-44.

Jackson, T. & Taylor, P. (1992) The precautionary principle and the prevention of marine pollution. *Chemistry and Ecology*, **7** (1-4): 123- 134.

Janzen, D.H. (1979) How to be a fig. *Annual Reviews in Ecology and Systematics,* **10**: 13-51.

Johnston, P.A. & Simmonds, M.P. (1991). Green light for precautionary science. *New Scientist,* August 3rd 1991.

Kirchner, J.W. and Well, A. (2000) Delayed biological recovery from extinctions throughout the fossil record. *Nature,* **404**: 177-179.

Laurance, W.F. & Bierregaard, R.O. (1997) *Tropical Forest Remnants: Ecology, Management and Conservation of Fragmented Communities.* University of Chicago Press, Chicago. 616pp

Laurance, W.F., Delamônica, P., Laurance, S.G., Vasconcelos, L and Lovejoy, T.E. (2000) Rainforest fragmentation kills big trees. *Nature,* **404**: 836.

McCann, K.S. (2000) The diversity-stability debate. *Nature*, **405**, 228-233.

Moyle, P.B. & Moyle, P.R. (1995) Endangered fishes and economics: intergenerational obligations. *Environmental Biology of Fishes* **43**: 29-37

Myers, N. (1993) Biodiversity and the precautionary principle. *Ambio*, **22**, 74-79.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonesca, G.A.B. & Kent, J. (2000) Biodiversity hotspots for conservation priorities. *Nature,* **403**: 843-858.

Noss, R.F. & Cooperrider, A.Y. (1994) *Saving Nature's Legacy: Protecting and Restoring Biodiversity.* Island Press, Washington DC, 416pp.

Oksanan, M. (1997) The moral value of biodiversity. *Ambio,* **26** (8): 541-545

O'Niell, J. (1997) Managing without prices: the monetary valuation of biodiversity. *Ambio,* **26**(8): 546-550

Power, M., & McCarty, L.S. (1997) Fallacies in ecological risk assessment practices. *Environmental Science and Technology,* **31:** 370A-375A

Reschler, N. (1998) *Predicting the Future: An Introduction to the Theory of Forecasting*. State University of New York Press, Albany NY. 315pp

Sala, O.E., Chapin III, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M. & Wall, D.H. (2000). Global biodiversity scenarios for the year 2100. *Science,* **287**: 1770-1774.

Sandin, P. (1999) Dimensions of the precautionary principle. *Human and Ecological Risk Assessment,* **5**(5): 889-907

Santillo, D. & Johnston, P. (1999a) Is there a role for risk assessment within precautionary legislation? *Human and Ecological Risk Assessment,* **5**(5): 923-932

Santillo, D. & Johnston, P. (1999b) Ethical standards and principles of sustainability. In: Addo, M.K. (ed.) *Human Rights Standards and the Responsibility of Transnational Corporations,* Kluwer Law International, The Hague, Netherlands, pp.351-368.

Santillo, D. Stringer, R.L., Johnston, P.A. & Tickner, J. (1998) The precautionary principle: protecting against failures of scientific method and risk assessment. *Marine Pollution Bulletin,* **36**(12): 939-950

Santillo, D., Johnston, P., & Stringer, R.L. (1999). The precautionary principle in practice: a mandate for anticipatory preventative action. In: Raffensperger, C. & Tickner, J. (eds.) *Protecting Public Health and the Environment: Implementing the Precautionary Principle,* Island Press, pp36-50

Sherden, W.A. (1998) *The Fortune Sellers: The Big Business of Buying and Selling Predictions* Wiley, New York 308pp.

Stairs, K.C. & Johnston, P.A. (1991) *The precautionary action approach to environmental protection.* Proceedings of the International Conference on Environmental Pollution, Lisbon. Volume 2. Inderscience, Geneva. pp473-479

Stirling, A. (1999) *On Science and Precaution in the Management of Technological Risk,* Volume 1: A synthesis report of case studies. European Science and Technology Observatory

(ESTO) Project Report for the European Commission Joint Research Centre, Institute for Prospective Technological Studies, Sevilla, Report No. EUR 19056 EN, 77 pp.

Suter, G.W. (1994) Endpoints of interest at different levels of biological organisation. In: Cairns, J. & Niederlehner, B.R. (eds.) *Ecological Toxicity Testing: Scale Complexity and Relevance* pp. 35-48. Lewis Publishers, Boca Raton.

Tickell, C. (1997) Foreword: the value of biodiversity. In: Ormond, R.F.G., Gage, J.D. & Angel, M.V. (eds.) *Marine Biodiversity: Patterns and Processes.* Cambridge University Press.

WRI World Resources Institute (1998) *World Resources 1998-1999 a guide to the global environment.* Oxford University Press, Oxford.