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ASSESSING FOREST INTEGRITY AND NATURALNESS IN RELATION TO BIODIVERSITY

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The Forest Resources Assessment Programme

Forests are crucial for the well being of humanity. They provide foundations for life on earth through ecological functions, by regulating the climate and water resources and by serving as habitats for plants and animals. Forests also furnish a wide range of essential goods such as wood, food, fodder and medicines, in addition to opportunities for recreation, spiritual renewal and other services.

Today, forests are under pressure from increasing demands of land-based products and services, which frequently leads to the conversion or degradation of forests into unsustainable forms of land use. When forests are lost or severely degraded, their capacity to function as regulators of the environment is also lost, increasing flood and erosion hazards, reducing soil fertility and contributing to the loss of plant and animal life. As a result, the sustainable provision of goods and services from forests is jeopardized.

FAO, at the request of the member nations and the world community, regularly monitors the world's forests through the Forest Resources Assessment Programme. The Global Forest Resources Assessment 2000 (FRA 2000), reviewed the forest situation by the end of the millennium. FRA 2000 included country-level information based on existing forest inventory data, regional investigations of land-cover change processes and a number of global studies focusing on the interaction between people and forests. The FRA 2000 Main report published in print and on the World Wide Web in 2001.

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Executive Summary

1) Efforts to assess and monitor forests, which previously focussed primarily on area and timber supply, are being expanded to reflect the full range of goods and services that forests provide, including the preservation of biological diversity.

2) In providing habitat for more than half of the world's species, forests play a major role in maintaining global biodiversity. Species richness varies among forest types and locations, but the capacity of any given forest to retain its original complement of biodiversity is a crucial factor in biodiversity trends.

3) Human activity is affecting the capacity of forests for biodiversity preservation through reduction in overall forest area (deforestation), changes in the spatial configuration of forests (forest fragmentation) and changes to forest structure and composition. While the first of these is already being monitored at global scales, this paper proposes approaches for monitoring changes in forest configuration at similar scales and for evaluating the magnitude of human influence as an indicator of other ecological changes

4) The impacts on biodiversity of alteration of forest spatial configuration by deforestation and fragmentation are primarily through:

   - Area effects – the tendency of small forest patches to support only subsets of the biodiversity complement of large areas and to be more vulnerable because of their size;
   - Edge and gradient effects – the impact of the interface with non-forest ecosystems, which affects environmental variables and biotic interactions;
   - Isolation effects – the separation of populations of forest organisms from similar populations and other forest areas, reducing genetic change and diversity and resource availability.

5) Measuring forest configuration and spatial integrity at broad geographic scales can be done using geographic information systems (GIS) to quantify indices that address each of these impacts:
   - Patch size – the area of each contiguous unit of forest cover;
   - Spatially weighted forest density – the % of cells within a given radius that are occupied by forest;
   - Connectivity – the from each forest cell to 'core' forest distance along a forested route.

   These indices can all be presented in both mapped and statistical form to support decision-making. Constraints on their measurement include the coarse resolution of the land-cover data available for global and regional scale analyses.

6) A single summary index of *forest spatial integrity*, which combines the three basic indices is proposed as a useful indicator of forest capacity to retain a full biodiversity complement. It can be displayed in mapped form to support decision

making with biologically meaningful information. It can also be summarised statistically to provide a baseline for monitoring.

7) Other ecological changes brought about by human activity are better investigated by measuring the *driver* of change, human activity itself. This can best be assessed by combining spatial information about settlements, infrastructure and land use in relation to ecosystem distribution.

8) A well-developed example of this approach is the Australian Wilderness Index, which evaluates remoteness from human influence in terms of distance and land use intensity.

9) This approach is illustrated applied at the global scale, and proposed as the basis for a measure of forest naturalness. The *forest naturalness* index would be derived by overlaying the wilderness index with forest cover and assigning a wilderness index score to each forest unit.

10) The resulting data on forest naturalness could be displayed in both mapped and statistical forms, and baselines could be established for monitoring work.

11) The implementation of baseline assessments and subsequent monitoring of forest spatial integrity and naturalness as proposed in this paper would be a significant advance over current periodic forest assessments. It would ensure that they addressed biodiversity preservation as one of the multiple benefits included in the periodic assessment of the world's forest resources.

I Introduction

Until recently, efforts at assessing and monitoring forests have focused on the amount (area) of forest remaining and/or its standing timber volume. Increasingly the multiple benefits and functions of forests, including provision of non-wood forest products, hydrological functions, carbon sequestration and biodiversity protection have been recognised, and their consideration in forest assessment has been recommended. There is now a perception that not all forests are equal, whether it be for timber production or for conservation of biodiversity. The international community has placed a high priority on assessing and monitoring the capacity of forests to provide any given range of services and conveying this information to policy and decision-makers and the general public (Nyyssonen & Ahti 1996).

In this paper we discuss:

- ways in which the capacity of forests to maintain their original biodiversity can be assessed at broad geographical scales;
- the problems inherent in such assessments;
- the most useful ways of presenting the results, and
- steps that need to be taken to ensure effective monitoring in the future.

Forests and biodiversity

Biological diversity, or *biodiversity*, is an imprecise term that is used to refer to the diversity of life at the levels of genes, species and ecosystems, in a single locality or over broad geographic areas, including the Earth as a whole. Biodiversity is used to refer to not only the numbers but the types of genes, species and ecosystems existing in the area in question. There is usually an implicit assumption that naturally occurring or original components of biodiversity are of more value than articficially introduced or degraded ones, and therefore biodiversity preservation or protection is frequently understood to refer to the preservation of these qualities and components.

Forests play a significant role in maintaining the world's biodiversity. They provide habitat for more than half of the world's species (Groombridge & Jenkins 2000). A forest area contributes to global forest biodiversity by the number of species present and shared with other areas and the number of species it contains that are found nowhere else (endemics). The biodiversity complement of individual forests varies with forest type, and the climatic and soil factors that relate to it, as well as the biogeographic position and isolation of the forest. In general forests at low latitudes and low altitudes, with warmer and wetter climates, have higher species richness than those at high latitude or altitude, or with cooler and drier climates. Natural perturbations, such as storms and wildfires can also be important influences on forest biodiversity.

Natural global patterns in biodiversity are also altered by human action (Groombridge & Jenkins 2000). In the contemporary world, human activities may be the most important influence on forests' capacity to maintain their original biodiversity. Such activities as commercial and artisanal logging, large scale land conversion, fuelwood and charcoal production, slash and burn agriculture, harvesting of non-timber forest products, hunting and mining all affect forest biodiversity. Climate change resulting from modification of the atmosphere by anthropogenic emissions of carbon dioxide is also affecting the distribution and status of forest biodiversity.

Human impacts on forest biodiversity

Each of these types of human influence affects forests differently, and the magnitude of the effects will depend strongly on the methods employed locally, the forest type, and on other factors within and around the ecosystem. For example, commercial logging in temperate forests is often by clear cutting, which entirely removes forest cover in some areas and fragments remaining forest cover. In contrast, commercial logging in tropical forests is usually by selective felling which disrupts canopies and forest structure and alters species composition, but may not alter total forest cover or its spatial configuration. Secondary effects of logging such as increased access and resultant hunting are also important determinants of the status of forest biodiversity and the prospects for its preservation. Small-scale timber extraction differs yet again in its effects on forest condition. The effects of climate change are less localised, and are only beginning to be recognised. Thus, many factors influencing forest biodiversity are affected in varying and complex fashions by human activity.

In general human activities tend to affect any of three major aspects of forests:

1) The total area of forest remaining – many of man's activities remove forest cover either temporarily or permanently. Some forest types may disappear locally, and reduction in the total amount of habitat is a significant pressure on some forest species that can lead to local extinction.
2) The configuration of remaining forest cover – reduction in forest area is often accompanied by division of remaining forest cover into fragments, rather than continuous blocks. Forest biodiversity is affected by the consequent local reduction in habitat area, by the exposure of forest edges to new environmental and biotic influences and by isolation from other forest areas (more detailed discussion below).
3) The structure and composition of remaining forest – some human activities alter canopy structure, or focus disproportionately on particular species and specific components of their populations.

Constraints on evaluating forest capacity for biodiversity preservation

Thus, useful measures of forest capacity for biodiversity preservation are likely to address the amount of forest remaining, its configuration or integrity and its 'naturalness', or lack of anthropogenic disturbance. However, developing measures of disturbance and making them operational is not straightforward. Precise histories of forest disturbance and its intensity are rarely available, especially over broad geographical scales (Kapos & Iremonger 1998). The problem of describing and measuring human-induced change in ecosystems has been summed up by Groombridge (1992, p. 250) in a review of habitat/ecosystem classification:

> . . . Just as it is impossible to define rigidly the limits of any given ecosystem or habitat, so it is impossible to determine how much a given area of ecosystem or habitat has to change before it can be considered destroyed or converted. The problem is compounded by the fact that the natural environment is not static but rather dynamic, sometimes highly so, on a time scale ranging from hours to millions of years. It is thus difficult even to define an undisturbed ecosystem or habitat as a standard against which to measure degree of disturbance.

There is a wide range of problems to be grappled with in successfully formulating methods for measuring the impact of humans in forest ecosystems. Difficulties arise from:

- Different interpretations of the basic form and function of ecosystems – Some concerns focus on ecosystem processes, while others emphasise composition. Such different conceptual approaches may lead to very different conclusions as to appropriate sets of information for the description and measurement of change (O'Neill *et al.,.* 1986).
- Sensitivity to observation and process scale – The scale at which ecological phenomena are observed and measured will have a major bearing on the conclusions drawn (Goodall 1974; Allen *et al.,. 1*987; Allen and Hoekstra 1990; 1992; Noss 1991), and the uses to which they can be put.
- Ambiguity in identifying benchmark conditions – The notion of naturalness depends on a clear distinction between the presence and impact of human activity and natural ecological patterns and processes. This can be problematic if, as in much of Europe, there is little or no reference forest with little human influence to provide a basis for comparison
- Establishing whether human-induced change represents a fundamental shift in organisation or change within normal limits of forest dynamics and ecosystem processes;
- Uncertainty regarding the place of humans in the environment.

These difficulties pose significant technical problems and can give rise to potentially contradictory answers to questions concerning ecological change.

Two principal strategies can be adopted to address these issues and assess the naturalness and ecological integrity of forest ecosystems at broad geographic scales in biodiversity-relevant terms:

1. Use *indicators* to assess key aspects of forest structure and/or function. This strategy has the obvious advantage of directly addressing the primary concern, which is the state of the ecosystem, but is subject to the difficulties described above concerned with the defining and measuring human-induced change in ecosystems. Attention must focus on parameters that can be evaluated in a globally consistent manner at broad geographical scales and are clearly related to the status of component biodiversity within the forests. Fragmentation of forests and parameters that describe it are a promising avenue for developing such indicators.

2. Measure the driver of ecosystem change - human activity, rather than the response - ecological change. This approach avoids fundamental problems associated with describing and measuring ecosystem response to human activity. Separation of the driver (cause) and response (effect) components also allows for greater precision and flexibility in analysis. General *indicators* of naturalness or ecological integrity can be developed on the assumption that the greater the amount of ecosystem exposure to human activity the greater the potential for human intervention in these ecosystems.

Indicators are measurements that convey information about more than just themselves. They provide means for quantifying and simplifying information on complex issues. They are purpose-dependent, almost always open to various interpretations, and never tell the whole story. Indicators are needed because assessing and monitoring everything is impossible and because what ***is*** known needs to be conveyed to non-experts in policy-relevant form.

Good indicators are:

- *scientifically valid*, i.e. they relate appropriately to what they are supposed to represent;
- based on easily available data;
- responsive to change;
- *easily understandable*;
- *relevant to focal issues and users' needs*;
- *subject to target or threshold setting*.

This document presents some approaches to generating indicators of forest condition in relation to biodiversity that could be used to conduct a globally consistent assessment. It focuses primarily on indicators relating to forest fragmentation and exposure to human activity.

II Assessing human impacts on forest biodiversity

Forest area

Of the three types of human impacts on forests mentioned above, the only one for which a consistent effort at assessment has been made at regional or broader scales is the change in forest area. Since the 1980s FAO has provided statistical data on forest cover (FAO 1989, 1995), based on forest inventory, national reporting and high resolution remote sensing. Since the mid-1990s a geographic overview of the current distribution of forest cover has been available (WCMC 1996, 1997). Thus it is possible to provide statistical and mapped data about the amount of forest cover remaining, and to a lesser degree, about the amounts of particular types of forests and trends in the amount of cover. The greater the level of detail about forest types available within such data sets, the more relevant to issues of the conservation of forest biodiversity they will become.

Forest configuration

The changes in forest configuration that accompany changes in land use and forest area can have substantial effects on the capacity of forest ecosystems to maintain their original biodiversity. As forest ecosystems are divided into smaller patches, there are numerous effects on their biota, and the responses may vary substantially among species and among forest types. The extensive literature on the effects of forest fragmentation suggests that the effects can be broken down into three major groups: area effects, edge effects and isolation effects. What follows is a brief summary of characteristic components of these effects.

Area effects

When large forest blocks are broken into smaller ones, not all species are included in the remaining patches, simply because of sampling effects (Wilcox 1980). This is especially true for rare species and for non-mobile organisms, such as trees and many invertebrates, which may be sparsely or patchily distributed within the forest. Large animals and top carnivores are well known to require large areas of habitat. These species are especially vulnerable to the reduction in habitat area caused by forest fragmentation, and they may disappear entirely from forest patches because food or other resources are inadequate to support them (Rylands & Keroughlian 1988, Soulé *et al.*, 1979, Schaller & Cranshaw 1980, Newmark 1987, Laurance *et al.*,. 1997b). Even smaller species are affected by the size of forest patches; amphibian species richness increased logarithmically with patch size in forest remnants in Madagascar (Vallan 2000). The disappearance of some species from forest fragments can profoundly affect the forest itself, as shown by the effects on tree communities of the disappearance of seed-eating rodents from forest islands in Gatun Lake in Panama (Putz *et al.*, 1990). Other species persist, but in smaller populations, which may encompass less genetic diversity and lead over time to the vulnerability of those species to other ecological changes such as disease. Rare species and those that normally occur at low population densities are especially vulnerable to these effects (Laurance *et al.,* 1997b). Smaller forest patches may also include less environmental variability and therefore fewer microhabitats than more extensive forest areas. This can contribute to the loss of individual species and may cause a reduction in the total species richness per area of forest (Scariot 1999). Fragmentation of forest cover may also alter the nature and proportional impact of natural disturbance regimes and regeneration processes (Laurance *et al.,*. 1997a, Viana *et al.,*. 1997).

Edge and gradient effects

Another important effect of forest fragmentation results from the creation of interfaces with non-forest environments. These interfaces are associated with environmental gradients resulting from the exposure of the forest edge to drying winds and increased solar radiation (Kapos 1989, Camargo & Kapos 1995, Kapos *et al.,*. 1997). The physical gradients affect ecological processes including canopy gap formation (Kapos *et al.,*. 1997), biomass and nutrient cycling (Laurance *et al.,*. 1997a, 1998a,b, Sizer *et al.,*. 2000), regeneration (Benitez-Malvido 1998, Sizer & Tanner 1999) and predation (Keyser *et al.,*. 1998) that can profoundly affect native species. Invasive species, both native and non-native, are often favoured by an increased incidence of forest edges within the landscape, so that substantive changes in species composition have been documented in forest fragments (Brown & Hutchings 1997, Laurance *et al.,*. 1997b, Lynam 1997, Malcolm 1997, Viana *et al.,*. 1997). Although some ‘edge effects’ have historically been regarded positively, principally because many game species make use of forest edges, they are generally regarded as detrimental to most native forest species. The magnitude of edge effects within forest fragments can be strongly affected by the land-cover characteristics in the surrounding landscape, the matrix harshness (Murcia 1995, Laurance *et al.,*. 1997b), and they are also dynamic, frequently increasing in magnitude and extent over time (Gascon *et al.,*. 2000). Connections among habitat fragments are an important means of reducing genetic isolation and providing additional foraging and refuge areas (Saunders *et al.,*. 1991)

Isolation effects

The other major group of effects of forest fragmentation results from the separation of the forest fragments from each other and from larger blocks of forest. This isolation reduces the movement of species that are reluctant or unable to cross non-forest areas and for those that depend on such species for dispersal. Reduced movement and dispersal also increases the chance of local extinction of individual species as a supply of colonisers or propagules is lacking. Isolation of fragments may also reduce the genetic neighbourhood of some trees, reducing the breadth of the local gene pool for cross fertilisation (Nason *et al.,*. 1997).

Responses to all of these effects vary among species, but a body of empirical evidence is accumulating that facilitates predictions about the likely effects of fragmentation on any particular forest ecosystem. Also from such evidence, a series of empirical generalisations concerning the spatial configuration of habitat with respect to biodiversity preservation can be paraphrased as follows (Noss & Cooperrider 1994; Thomas *et al.,* 1990).

i. Habitat that is more widely distributed across its original range is more likely to persist than habitat confined to small parts.

ii. Large blocks of habitat are superior to small blocks of habitat.

iii. Blocks of habitat close together are better than blocks far apart.

iv. Habitat in contiguous blocks is better than fragmented habitat.

v. Interconnected blocks of habitat are better than isolated blocks.

The long-term maintenance of forest integrity depends on promoting these characteristics in landscapes. The more that landscapes retain these characteristics, the less is their vulnerability to human-induced change. These generalisations provide a useful basis for assessment and communication of information about forest status in this respect.

Human activity

Forest structure and composition and their implications for biodiversity are difficult to evaluate at broad geographic scales and may vary widely depending on (among other factors) the kinds and intensity of human activity and local ecological conditions. Therefore, assessing the amount of human activity may be a useful proxy for evaluating its impacts on biodiversity. It is well-documented, for example, that logging increases the probability of recurrent fire in Amazonian rain forests (Uhl *et al.,*. 1991, Nepstad *et al.*, 1998), and that this in turn will lead to long term changes in species composition (Cochrane & Schulze 1998, 1999). Logging also affects animal community composition and ecological relationships (e.g. Johns 1996, Lambert 1992, Ochoa 2000). Hunting activity near settlements substantially reduces the abundance of mammal species (Muchaal & Ngandjui 1999) and the construction of roads facilitates both logging and hunting as well as land conversion and colonisation.

III Measuring forest configuration and spatial integrity

Source data

The evaluation of forest cover configuration depends on the availability of spatially referenced or mapped data. Most commonly, such data are derived from remote sensing, using either satellite-mounted or airborne sensors or aerial photography. For continental and global scale evaluation, satellite imagery is the most appropriate data source.

Satellite-mounted sensors measure radiation reflected from the Earth's surface in a variety of spectral bands; different land-cover types have different characteristic reflectances and spectral signatures. Satellite sensors vary in the frequency with which they return to a given portion of the earth's surface (temporal resolution), in the spatial resolution, or pixel size, of the data they provide, and in the spectral resolution or numbers and types of spectral bands in which data are recorded.

Satellite data are processed by adjusting them radiometrically to compensate for variations in atmospheric conditions, and then classifying the digital data by one of several methods. Classification can be achieved by plotting reflectance in particular spectral bands or band ratios such as NDVI (Normalised Difference Vegetation Index) and visually interpreting the results. Alternatively, satellite data can be classified digitally by grouping pixels with similar spectral characteristics, and by comparing their spectral responses to those of pixels from areas of known land cover (supervised classification). Additional power can be brought to the process by incorporating information on variation in spectral response of an area through time (seasonal and other changes) and ancillary data (on land forms, land use etc.). Expressing pixel composition in relation to mixtures of the spectral responses of possible component land covers (spectral mixture modelling) can also improve the resolution and accuracy of classification.

The processing of satellite data is expensive and time consuming, requiring sophisticated hardware and software to deal with the large volumes of digital data involved. Although very sophisticated and spatially detailed vegetation maps have been generated from satellite data, these have been confined to small areas because of the additional volume of data required for high spatial and spectral resolution processing.

The only currently available global land-cover data set derived from satellite data that have been processed in a consistent manner is the GLCCD, produced by the EROS Data Center and IGBP, from monthly averages of data from the AVHRR satellite during 1992-93 (Belward *et al.,*. 1999). These data have a spatial resolution of 1 km, and their relatively low spectral resolution has been compensated by the large amount of data on temporal variation that is available and by incorporating large amounts of ancillary data into the classification process (Loveland *et al.,*. 1999). At present, this is the only data set that could be used to evaluate forest fragmentation and regional and global scales.

Because of the coarse spatial resolution of the GLCCD, the data represent relatively coarse spatial mosaics and provide insufficient detail for certain kinds of analyses of forest configuration. There are also systematic errors in that result from the coarse spatial resolution: small non-forest patches in areas of high forest cover remain undetected as do small forest patches in landscapes with low forest cover. These phenomena lead respectively to over- and under- estimates of forest cover. Although these errors can be reduced by calibration against high-resolution data (Mayaux & Lambin 1995, 1997), it is not clear whether such a process has been applied to the GLCCD. Details of boundary configuration and interspersion of forest and non-forest land cover are lost at the lower spatial resolution. However, some consistent patterns that have ecological meaning for forest biodiversity do emerge. The following discussion focuses on the options for evaluating forest configuration that are appropriate for use with data like the GLCCD, but also includes examples of metrics that can be used productively with higher resolution data. The sensitivity of the different measures to data scale and resolution is discussed.

Other constraints

Defining ecosystems is an issue in the extent to which the analysis attempts to focus on individual forest types and their configuration within the landscape. Ideally one would look at the fragmentation parameters of each forest type within a mixed matrix of forest and non-forest separately, but in fact it is unlikely that the source data on forest type distribution can support this. It will probably be necessary to look just at forest cover in relation to the non-forest matrix, or perhaps at the fragmentation properties of rather broad or regional forest types with a minimum of overlap.

Scale is of course an issue. Fragmentation is differently determined for different components of the biota of a forest ecosystem. A path or deforested strip of a few metres width may be a significant barrier to an invertebrate, whereas a deforested strip of several km presents little obstacle to a forest-dwelling bird of prey. The data available for doing a globally consistent evaluation of forest fragmentation are 1 km resolution satellite data. Therefore, any fragmentation metrics that are derived from these data will represent the distribution of forests only at this coarse scale, and indeed might be better said to represent the configuration of forested areas in the landscape than of individual patches of forest. The effects of coarse resolution data on forest area estimates discussed above also apply to the estimation of forest fragmentation. Because of the problems of detecting small patches within very high or very low forest cover landscapes, fragmentation will be underestimated in areas of both very high and very low forest cover.

The question of what is natural or a *baseline* condition is also an issue in evaluating fragmentation of forests. Some forest types and regions are naturally more continuous than others. For example, forest close to latitudinal or other limits of its distribution has a tendency to be naturally patchy, as in the taiga/tundra transition of the boreal regions and some forest-savanna boundaries. Similarly forests that exist on mountains in non-forested landscapes because of the greater humidity at higher altitudes are inherently restricted in area and isolated from each other. The fragmented nature of these systems has a great significance for biodiversity that is entirely distinct from that of the fragmentation of continuous forest cover by human activity.

Some level of forest fragmentation may also result from natural disturbance and dynamic processes within forests. However, the scales of these processes are usually such that they will be beneath the resolution of any global or regional scale analyses. Although some kinds of storm damage or wildfire impact may be at scales that could be detected by these analyses, the issue of whether changes in forest fragmentation parameters are within or outside the range of natural variation is unlikely to arise.

Analysis tools

A number of packages for use with geographic information systems (GIS) permit the analysis and characterisation of landscapes in terms of their patch composition, spatial relations and dynamics. One such package, FRAGSTATS (McGarigal and Marks 1995) is widely used for the description and analysis of landscape configuration. It offers a wide range of measures of varying complexity.

A proposed approach to measuring and monitoring forest spatial integrity

The choice of measures of forest fragmentation for use as indicators of forest capacity to retain biodiversity is dictated both by the source data and by the range of biological effects being targeted. Summary statistics of landscape metrics are of little use for predicting responses of individual species without more detailed information about both species requirements and environmental variation on the ground. However, to provide both an overview of forest status in relation to biodiversity and baselines to track changes that may affect forest biodiversity simple statistical expressions of forest configuration can be useful. It is also important that the metrics chosen be easily communicated and understood by the anticipated audience for the overview and monitoring, so conceptually complex indices are generally less useful.

Landscape ecological theory and GIS technology have generated a number of measures of the spatial distribution of habitat that express different aspects of its fragmentation in ways that relate to ecological processes. These have mostly been used in the evaluation of habitat and landscape processes at management unit scales. For example, Kramer (1997) used these analyses to describe landscape change as a result of management of two national parks in Costa Rica. She found that as forest cover increased, patch size increased and patch shapes became more compact for the shrinking pasture areas, but remained similar or became more complex over time for the expanding forest. Landscape diversity declined as the pasture areas shrank. In another example, Helmer (2000) used similar analyses to show that in the mountains of Costa Rica secondary forest is strongly associated with primary forest and occurs in smaller patches with more complex shapes. Logsdon *et al.*,. (2000) used

FRAGSTATS to analyse and describe landscape composition, configuration and heterogeneity from remotely sensed data for a small part of central Amazonia. In one of the few studies that directly addresses the relationship between landscape parameters and potential management actions, Ranta *et al.,*. (1998) used landscape analysis to characterise the fragmentation of Atlantic rain forest in Brazil and to simulate the likely impacts of land use change or forest restoration. Still other studies have characterised forest fragmentation at broader scales. Skole & Tucker (1993) evaluated forest fragementation and potential edge impacts for the whole of the Amazon Basin, and substantial research has been done to characterise and describe patterns of fragmentation throughout the tropics (Jeanjean *et al.,*. 1994, 1995).

As these examples show, spatial analysis of landscapes and the metrics it produces have been the subject of much research and have been used widely for descriptive purposes. However, they represent a complex suite of ecological processes and effects that are beyond the detailed understanding of many decision-makers. Also, they often focus at scales (such as the 100 m edge effect postulated by Helmer 2000) that are beyond the resolution of currently available classified satellite data on landcover at regional and global scales. Furthermore, they have rarely been used to identify the likely relative value of different patches of habitat in the context of fragmentation, as a basis for monitoring changes in that condition or for making policy and management decisions.

Thus, there is a need for an easily understood summary index that can be used both as a basis for visualising the relative biodiversity preservation capacity of different forest areas and to establish a baseline for tracking forest landscape change. Both of these uses support decision-making and evaluation of policy effectiveness. Such an index needs to reflect all three types of fragmentation effects outlined above:

- area or patch size;
- interface with non-forest, or edge effects
- isolation from, or interconnection with other patches.

In this section of the paper, we develop a set of tools for quantifying forest fragmentation in relation to these three types of effects and displaying them in ways that can be meaningful to decision-makers. The use of each tool is illustrated using the forest cover of Paraguay (Figure 1) as an example data set. The data are part of a global data set derived from 1 km resolution AVHRR satellite data, and are therefore characteristic of forest cover data that could be used in global assessment and monitoring of forest fragmentation. Throughout the paper it will be useful to compare the mapped indices of fragmentation with this base map of forest cover to evaluate the additional information they convey and their potential utility for supporting decision making.

Forest Patch Size

Area effects are most easily represented in terms of patch size. A GIS can be used to identify all patches of forest within the area of study, to measure their areas and assign them to a patch size class. If the data used are coarse resolution satellite data, this evaluation will identify contiguous blocks where forest is the predominant land cover, even if there may be some breaks in the forest cover at sub-pixel level.

The appropriate size class intervals can be selected empirically to provide a distribution with an easily analysed shape. The thresholds can also be adjusted to suit different regional characteristics or to address specific conservation issues and values. For example, selection of patch size thresholds might be related to average individual home range size or sustainable micro-population area of forest animals. Other criteria might be based on the spatial scale of natural regeneration and successional processes that provide continuity of forest ecosystems after disturbance. However, it is important to recognise that consistency of classes between sampling times is an essential component of any monitoring or comparative work.

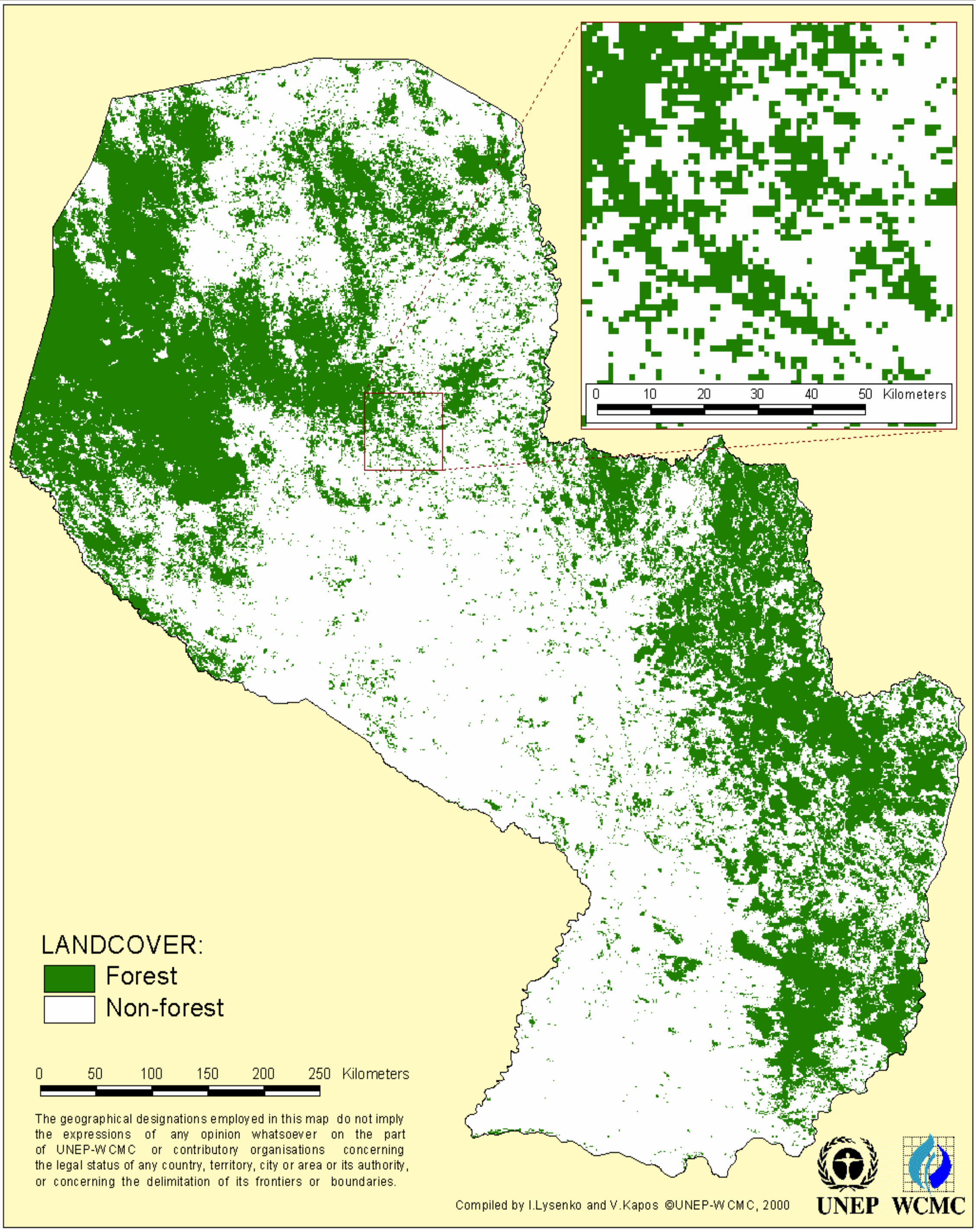

Figure 1. Forest Cover of Paraguay. This map, which shows the distribution of forest cover in 1 sq.km cells, is the basis for the analyses presented subsequently in this paper.

Data source: U.S.G.S. EROS Data Center/GLCCD Preliminary version (1997) South America Seasonal Land Cover Regions. The data, which have 1-km² nominal spatial resolution, are derived from 1992-1993 monthly AVHRR images analysed for NDVI

For the evaluation of forest fragmentation based on global datasets (mostly 1 km^2 pixel size), a conservative analysis would assume that details of forest patches smaller than 10 km^2 cannot be analysed reliably because of the high impact of variation in shape and sub-pixel level structure. It can also be accepted that all forest patches larger than 300 km^2 (30,000 ha) can be regarded as continuous forest, and thus make up the largest patch size class.

In the present analysis (Fig. 2), and potentially for future broad scale monitoring and data integration activities, logarithmic type scales are used, with the size class intervals delineated by rounded values in km^2. (This minor distortion relative to a truly logarithmic scale will not affect the majority of statistical tests that could be applied in comparing patch size distributions). When higher resolution data are available, the scale might be extended to split the smaller patch size classes, while still keeping the established pattern of scaling. Equally, the upper end of scale could be expanded for broad scale or global level studies.

For the purpose of developing an integrated index of fragmentation, numeric ranks ranging from 1 (1-10 km^2 patch) to 10 (> 300 km^2) were assigned to classes, and each 1 km^2 cell classed as forest was assigned to one of these ranks according to the size of patch it belongs to. The result (Fig. 2) is a visualisation of where forest occurs in large patches and where it occurs in small ones that may be clearer to non-experts than simple maps of forest cover. Such an analysis can also generate a statistical distribution of forest area among patch size ($\mathbf{P_s}$) classes that can be used as a baseline for assessment and monitoring of forest condition in relation to the capacity for biodiversity preservation (Fig 3; see section on presentation issues below).

A limitation of the patch size analysis is that it tends to identify barely connected and/or irregularly shaped patches of forest as belonging to larger size classes than may be appropriate in terms of their ecological function. For example, [A] and [B] in figure 2 indicate forest patches of comparable size that are classed differently because of the presence or absence of small connections (of the order of the resolution of the raster data) to larger patches. Given the characteristics of coarse resolution data and the implications of connections or breaks between forest patches at this scale, additional components are needed to improve the strength of the evaluation. Furthermore, the patch size analysis alone does a poor job of distinguishing between the capacities to support biodiversity of outlying narrow branches of patches and core areas of forest patches within the same size class.

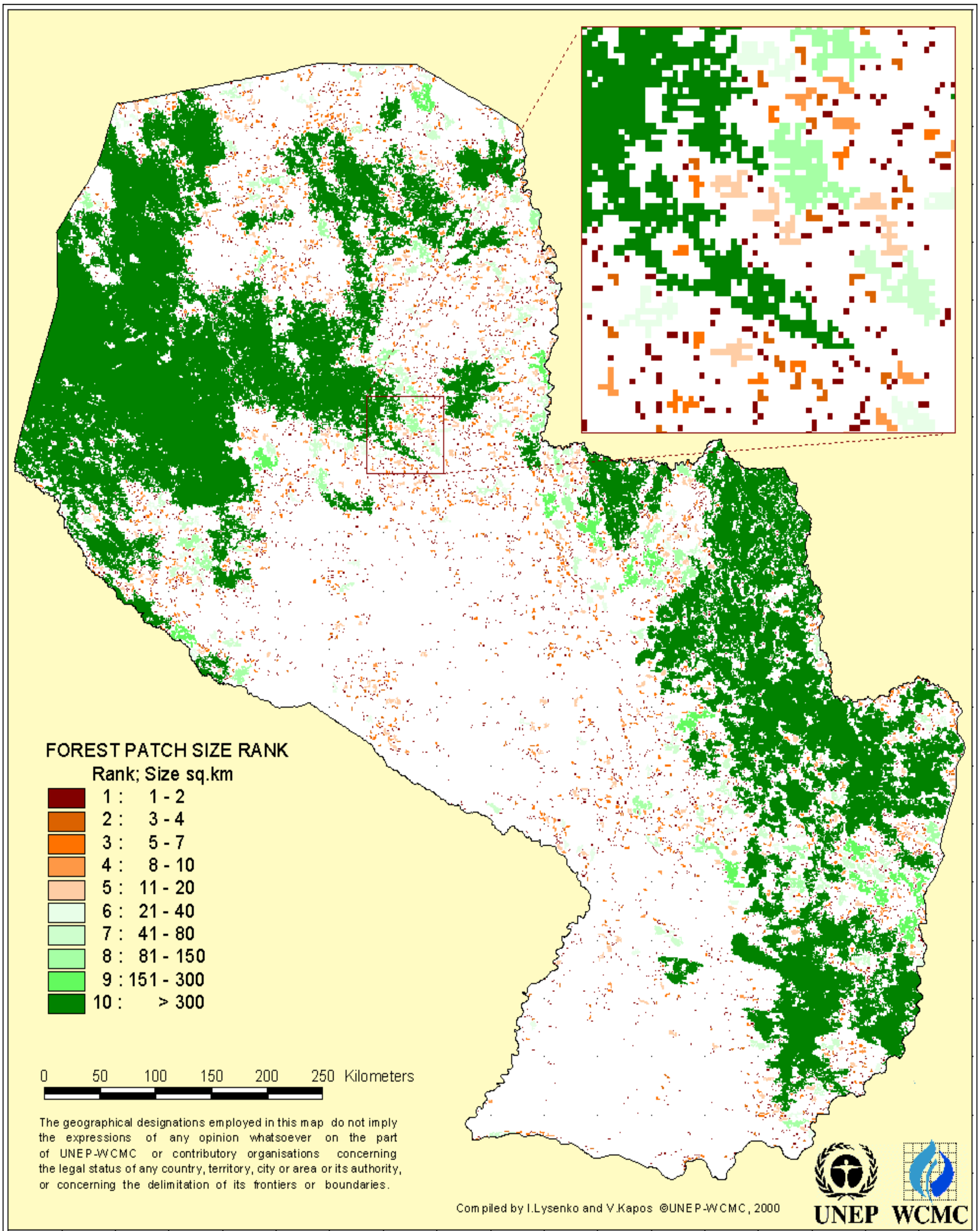

Figure 2. Distribution of forest cover in Paraguay (see Fig. 1) among different patch size classes.

The class intervals, which were selected empirically, could be altered to suit different regional characteristics or conservation value thresholds. For the present analysis, patches larger than 300 sq.km (30,000 ha) are regarded as continuous forest. Though such GIS analysis of patch size is a useful parameter, its interpretation is complicated by its tendency to identify barely connected and/or irregularly shaped patches of forest (e.g. [A] in the inset) as belonging to larger size classes than may be appropriate in terms of their ecological function. Similar sized patches are classed as small ones [B] if they are separated from other forest areas by small gaps comparable to the resolution (cell size) of the raster data.

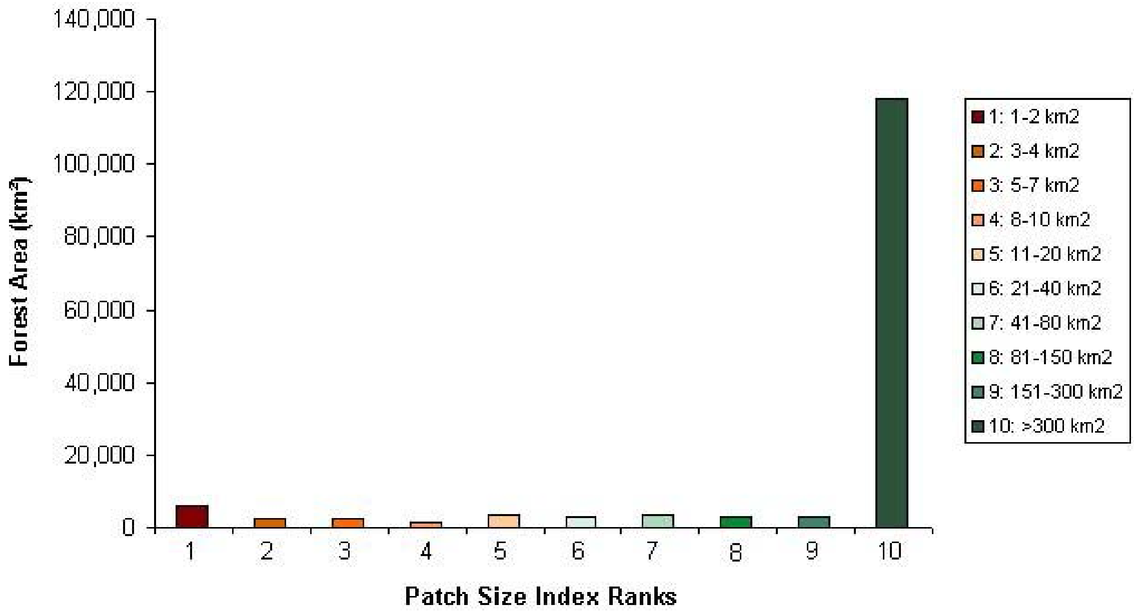

Figure 3. Statistical distribution of the forest area of Paraguay among different patch size classes. Such a distribution is an initial assessment of forest condition with respect to fragmentation and can be used as a baseline for monitoring purposes. If fragmentation of the forest increases, the amount of forest in the largest patch sizes will decrease and that in the smaller size classes will increase. However, it is common for deforestation to cause the disappearance of the smallest patches and a consequent reduction in the importance of that size class. Distributions can be compared between assessment times by using non-parametric statistical tests.

Shape or edge influence

The effect of the interface between forest and non-forest (edge effects) can be addressed through the relatively commonly employed shape indices, such as perimeter to area ratios and edge-to-core ratios. However, these measures are more appropriate to ground level studies or high resolution data sets in which the forest cover is real and the extent of edge influence is well understood. For coarse resolution data in which patches may or may not represent actually contiguous forest, an alternative approach is to evaluate the percentage of the neighbouring cells that contain forest within a given radius of each cell (Spatially Weighted Forest Cover Density – $\mathbf{S_F}$). The radius can be chosen in the light of known scaling issues and concerns about specific ecological phenomena. In the current illustrative example (Fig. 4) a radius of 5 km was chosen as being consistent with the spatial accuracy of the data.

As can be seen in Figure 4,the value of $\mathbf{S_F}$ is high when the sample cell belongs to the interior of, or is near a dense or solid patch that is comparable in size or larger than the radius used for calculation of $\mathbf{S_F}$. For areas at the periphery of, or distant from large continuous forest patches, $\mathbf{S_F}$ is lower and mostly dependent upon patch shape and isolation. Small isolated patches have small amounts of forest in their immediate neighbourhoods ([A] in Fig. 4); the extreme case is the single-pixel forest patch that is separated by more than the $\mathbf{S_F}$ evaluation radius from any other forest. Points along patch edges ([B]), and in patches of complex shape or in narrow strips of forest, where edge effects may influence forest status, also have low proportions of forest cover in their neighbourhoods, while points within larger patches [C] or continuous forest are entirely surrounded by forest. Therefore, spatially weighted forest density, $\mathbf{S_F}$ provides a basis for expressing the role of edge effects, without dependence on accurate definition of the edge or limit of a given patch.

Like the patch size analysis, this analytical approach can be presented in a way that provides a clear spatially referenced visualisation (Fig. 4). This shows where forests are subject to edge and isolation effects as distinct from forest that is both part of a large patch (that is predominantly forested) and distant from the interface between forest and non-forest. Although the implications of these effects for forest biodiversity are strongly dependent on forest type and location, **change** in the amount and location of forest subject to such influences is likely to result in changing biodiversity status. Therefore, establishment of both visual and statistical baselines for this parameter is an essential step in the monitoring of forest biodiversity status.

Spatially weighted forest cover density, $\mathbf{S_F}$, is measured directly in percentage units. In the present example, (Fig 4 and 5), the scale is broken into ten equal intervals and the lower percentages are assigned to lower ranks. This reflects an implicit assumption that forest habitat that is less subject to the influences of edges and isolation is of greater value for forest biodiversity. Although this assumption may be invalid in some cases, it is a realistic view at the global scale. As with the patch size distribution, appropriate breaks between classes can be selected on theoretical and empirical grounds, and a statistical summary of the forest area in each class can provide a basis for assessment and monitoring of forest condition in this context (Fig. 5). However, consistency in approach between assessments is a critical component of any monitoring or comparative analysis. The use of GIS tools that retain the source data and intermediate parameters in separate grids, provides flexibility and ensures that a consistent approach can be applied across all data sets in a time series.

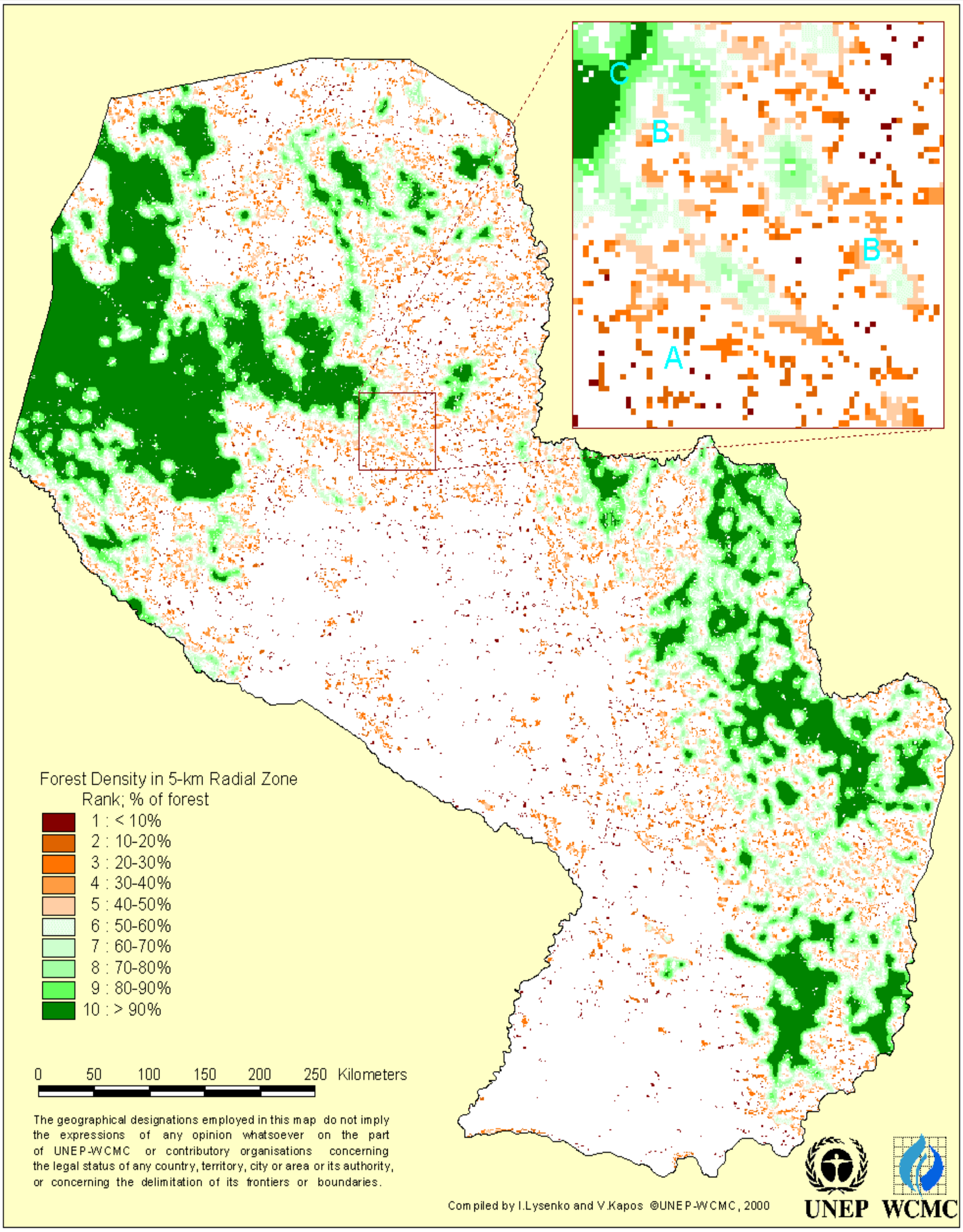

Figure 4. The forest cover of Paraguay displayed according to Spatially Weighted Forest Cover Density S_F, the proportion of cells within a 5 km radius that are forested.

The class breaks are equally distributed along the range from 1-100%. This measure partly reflects patch size, but it is also dependent upon patch shape and isolation. Small isolated patches have small amounts of forest in their immediate neighbourhoods [A]. Points along patch edges [B] and in narrow strips of forests also have low proportions of forest cover in their neighbourhoods, while points within larger patches [C] or continuous forest are entirely surrounded by forest.

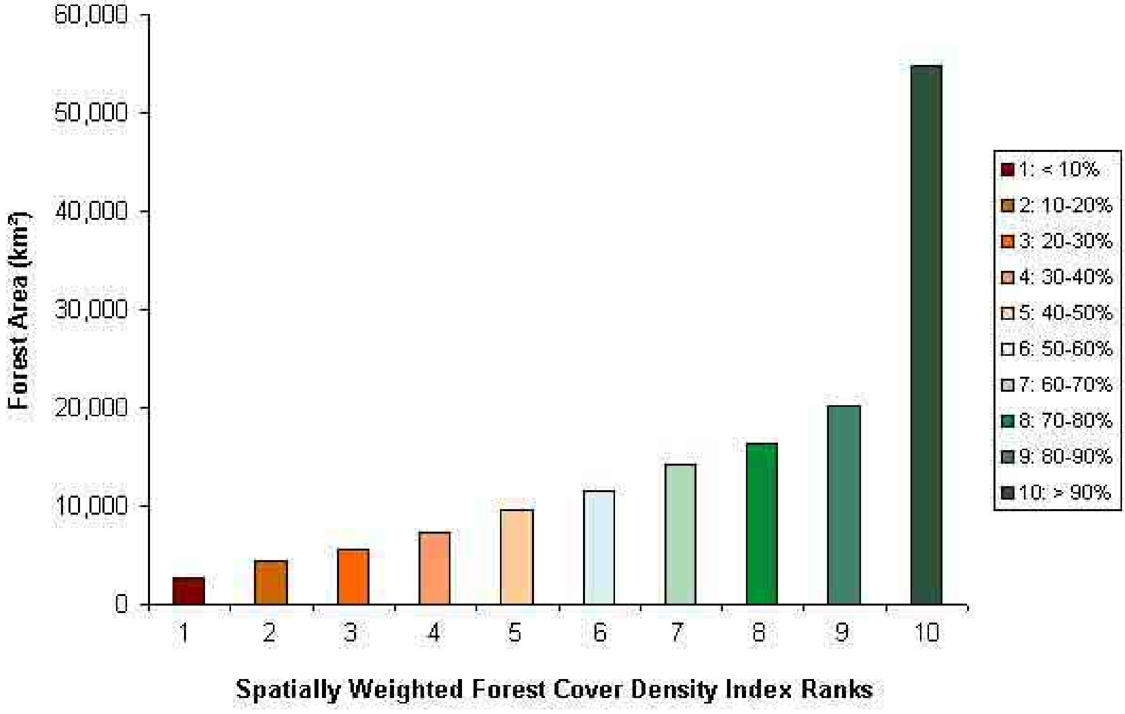

Figure 5. Statistical distribution of Paraguay's forest area among different classes of spatially weighted forest cover density, $\mathbf{S_F}$, forest occurrence within a 5 km radius. Such a distribution is an initial assessment of forest condition with respect to fragmentation, which reflects proximity to non-forested areas and edges, and can be used as a baseline for monitoring purposes. If fragmentation of the forest increases, the amount of forest in the highest classes (i.e. those forest cells that are completely surrounded by forest) will decrease, and that in the smaller classes will increase. Distributions can be compared between times using non-parametric statistical tests.

A simplified version of the forest cover density analysis can be used to identify forest density zones, generalised outlines of the forest areas that are likely to have the highest integrity, those of least integrity and the intermediate values (Fig. 6). This approach provides a useful way of defining "core" forest areas (see below). It also provides a means of focussing attention on the forest areas of intermediate 'quality' or integrity. These are the areas that may appear least distinct to the non-expert observer, but are most likely to be immediately affected by policy and management decisions. Separate focus on this zone of intermediate forest density can enhance the visibility of pattern and change in the statistical data (see section on presentation below).

Isolation and inter-connection

Quantification of the third component the effects of fragmentation, isolation, requires some measure of distance to other forest areas. However, the degree of isolation and/or the positive effects of interconnection are also dependent on the characteristics of the neighbouring forest. It is also true that many forest species are unable and/or reluctant to cross areas without forest cover (Laurance *et al.,*. 1997b), so forest areas that are directly connected to other forest areas are likely to be of greater value and more accessible to a greater range of forest species. The possibilities of dispersal to and from a particular forest area depend on the species of interest and the forest stand characteristics (among other factors).

Furthermore, the overall sustainability of biological systems is increased by the presence of relatively intact “core” areas surrounded by peripheral areas that are important in buffering the system as a whole against external impacts (see Fig. 6). These peripheral areas, in their turn, also benefit from connection to “core” areas, which provide necessary genetic resources (*via* animal migration or plant dispersal) that can be key to maintaining ecosystem function after natural or anthropogenic disturbance. Therefore, forest patches that are connected by forest to core forest areas may be viewed as more sustainable and of higher biological value that areas of similar forest cover density and overall shape that are not connected to core forest.

Core forest area, and connection to it, may be defined using thresholds appropriate to the particular components of biodiversity of interest, management considerations and properties of individual forest types. In the present study, for general illustration core forest area was defined using two criteria that are appropriate to the coarse scale of the data (Fig. 6):

1. Core area is represented by continuous forest with density ($\mathbf{S_F}$) more than 90%
2. The size of an individual core area must be at least 100 km^2

The distance to core forest areas, *via* cells containing forest cover, estimates the degree of connection of forest cells to core areas. The connectivity, $\mathbf{C_F}$, is inversely proportional to distance from core areas, ranging from 10 (core area) to 1 (24-27 km), and in this example, forest cells at distances greater than a threshold of 27 km from core forest are regarded as effectively isolated and assigned a $\mathbf{C_F}$ value of 0. Forest cells that are connected to core forest by between one and 27 km of forest are assigned to intermediate classes of connection (Figure 7).

Like the previous two indicators, this approach can be used to provide both a spatially referenced visualisation and a statistical summary (Figure 8) of which forest areas are likely to be in the best condition for preserving forest species. In this case, the best condition refers to the closest connection to core forest, and therefore the greatest accessibility to the greatest range of forest species. Although this approach is proposed for the analysis of relatively low resolution (1 km) data, where individual pixels may include mosaic patterns, and species diversity and successional stage may not be incorporated, the same principles would apply at more detailed scales.

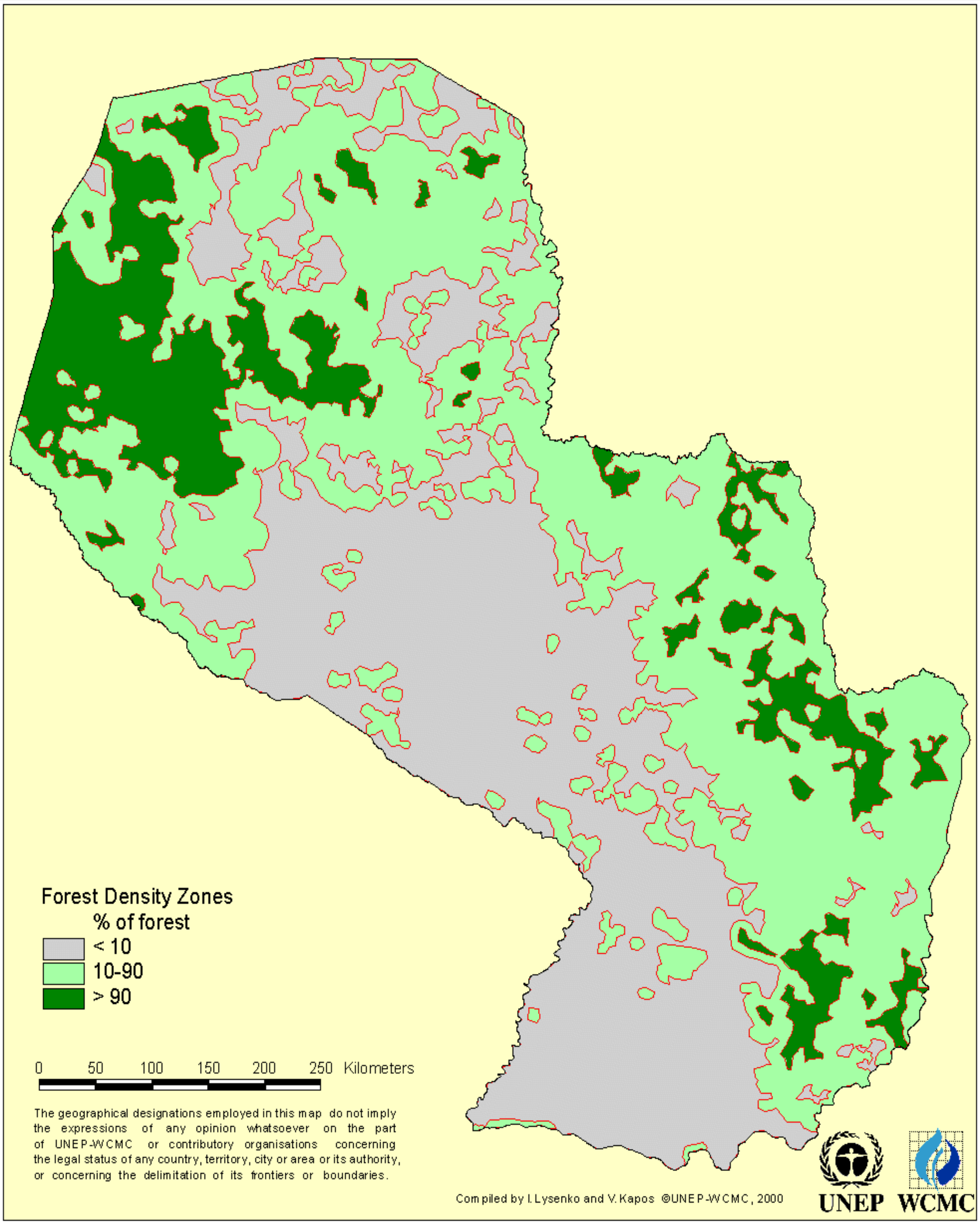

Figure 6. Spatially Weighted Forest Density Zones in Paraguay displayed according to the proportion of cells within a 5 km radius that are forested.

The zones are non- or low-forested (**S**$_F$ <10%) areas, intermediate (**S**$_F$ is in range 10-90%) and high density (**S**$_F$ >90% of forest) areas that can be considered as "Core Forest".

The outline of the zones is generalised to distinguish only patches above a minimum size of 100 sq.km.

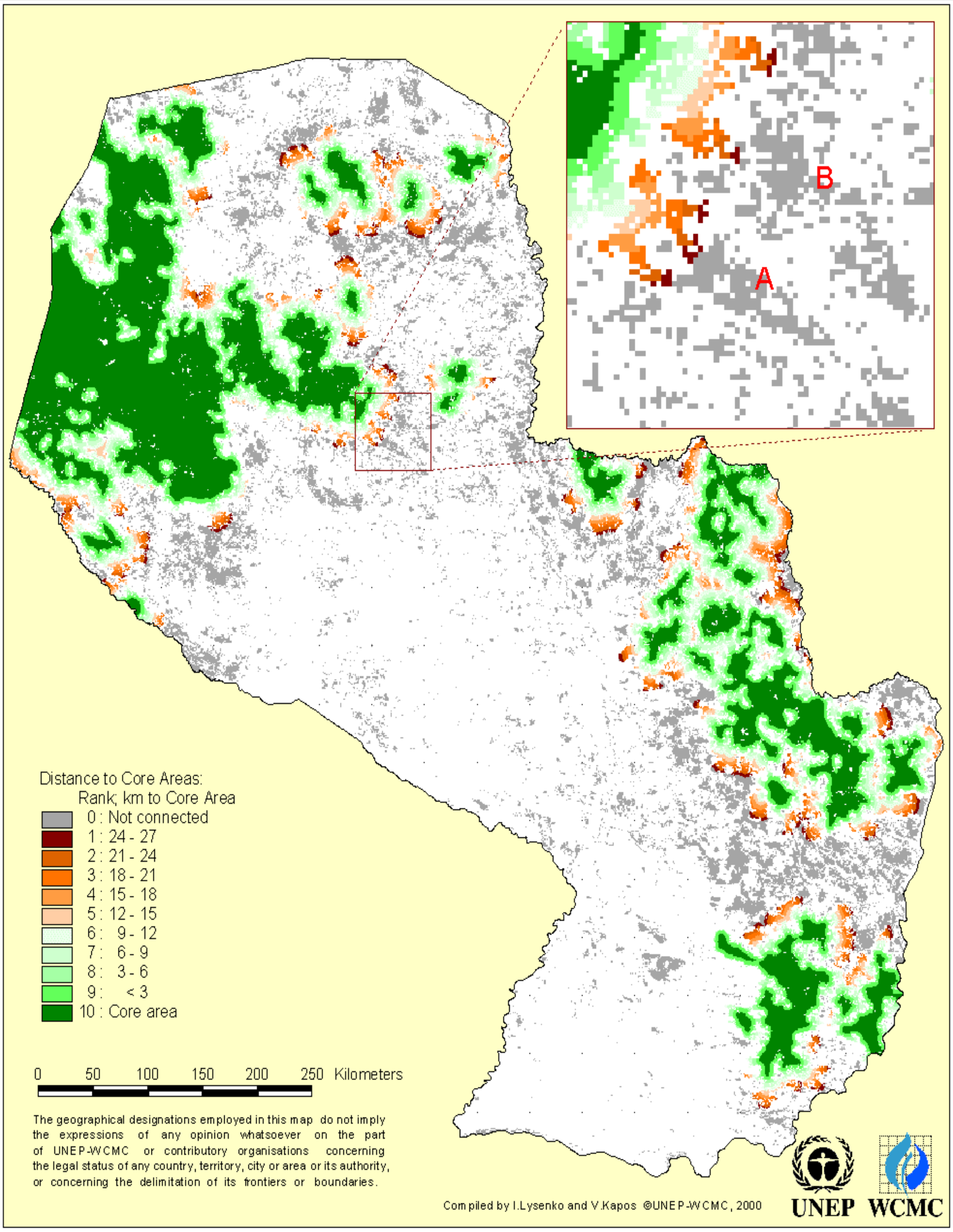

Figure 7. The forest cover of Paraguay displayed according to $\mathbf{C}_F$, the distance via continuous forest links to the nearest Core Forest Area.

Forest areas that are part of large patches, but more than 27 km from core forest areas, via forest [A] are classed as unconnected, as are those forest areas that are completely disconnected from large or core patches [B].

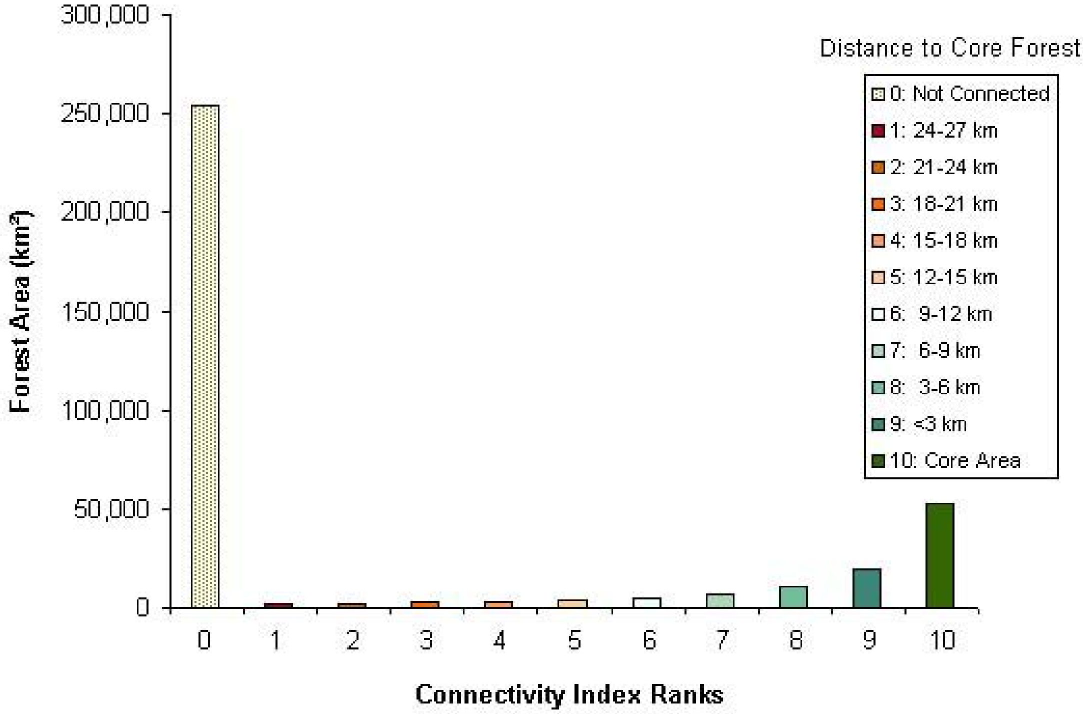

Figure 8. Statistical distribution of forest area in Paraguay among different classes of connectivity, $\mathbf{C_F}$. This distribution could serve as a baseline for monitoring the changes in forest capacity to retain biodiversity. As forest fragmentation increases, the amount of core and highly connected forest will decrease, and unconnected or remotely connected forest area will increase both absolutely and as a proportion of the total. Some scenarios of forest regeneration could result in increasing amounts of connected forest.

Forest Spatial Integrity Index

Despite the individual limitations of each of the above indices, between them they cover all three important aspects of forest fragmentation effects and make it possible to quantify most variations in the spatial distribution of forest cover. Patch size, P_s, facilitates comparison of forest stands larger than some minimal patch size threshold and below a size that can be regarded as continuous forest, regardless of variation. Spatially weighted forest cover density, S_F, provides a way of identifying both small, dispersed patches and areas subject to edge effects. However, it provides little detail at intermediate values and gives insufficient information about the interconnection of forest patches with forest patches of different status. The connectivity index, C_F, permits ranking patches of similar size in relation to their probable accessibility to forest species, and provides a way of distinguishing among forest areas of intermediate sizes and densities.

As all three indices range between 0 and 10, it is feasible to integrate them, using averaging or some more complicated method of combination, to provide a combined measure that is similarly easy to understand and interpret. The contributions of the individual indices can be adjusted using numeric coefficients to reflect the conceptual weight attached to the different factors:

$$\mathbf{F_F} = x(\mathbf{P_S}) + y(\mathbf{S_F}) + z(\mathbf{C_F})$$

As patch size and forest density are interrelated, while connectivity is conceptually distinct and potentially very important in sustaining forest biodiversity, we suggest the following composite index of forest spatial integrity as a starting point for analysis of forest condition:

$$\mathbf{F_F} = 0.25\mathbf{P_S} + 0.25\mathbf{S_F} + 0.5\mathbf{C_F}$$

Other coefficients could be adopted to reflect a different focus.

The example application of this forest spatial integrity index, $\mathbf{F_F}$, to the forest cover data for Paraguay (Fig. 9) demonstrates the visual impact and clarity provided by this approach, which could ensure its utility for decision making. The statistical distribution of forest area among the different integrity index classes (Fig. 10) provides a baseline for monitoring forest spatial integrity over time. Such monitoring is an essential component of evaluating policy effectiveness and the impact of management decisions.

Presentation of results

Great care must be taken in the presentation of results of such assessment and monitoring. Data in a mapped context may be the most useful for supporting site-specific decision-making and, conceivably, for scenario testing. Data in statistical form are potentially more useful for monitoring and evaluation of policy effectiveness. It is crucial that consistent methods are applied for comparisons in space and time and that original data and analyses are retained to permit reanalysis in the event that changes in thresholds or approaches are deemed appropriate.

Statistical data need to be presented in absolute areas rather than as percentages. Changes in percentages of forest cover in lower integrity categories may reflect either an improvement in the integrity of formerly low value areas, or simply a loss of forest area from those categories without an increase in others.

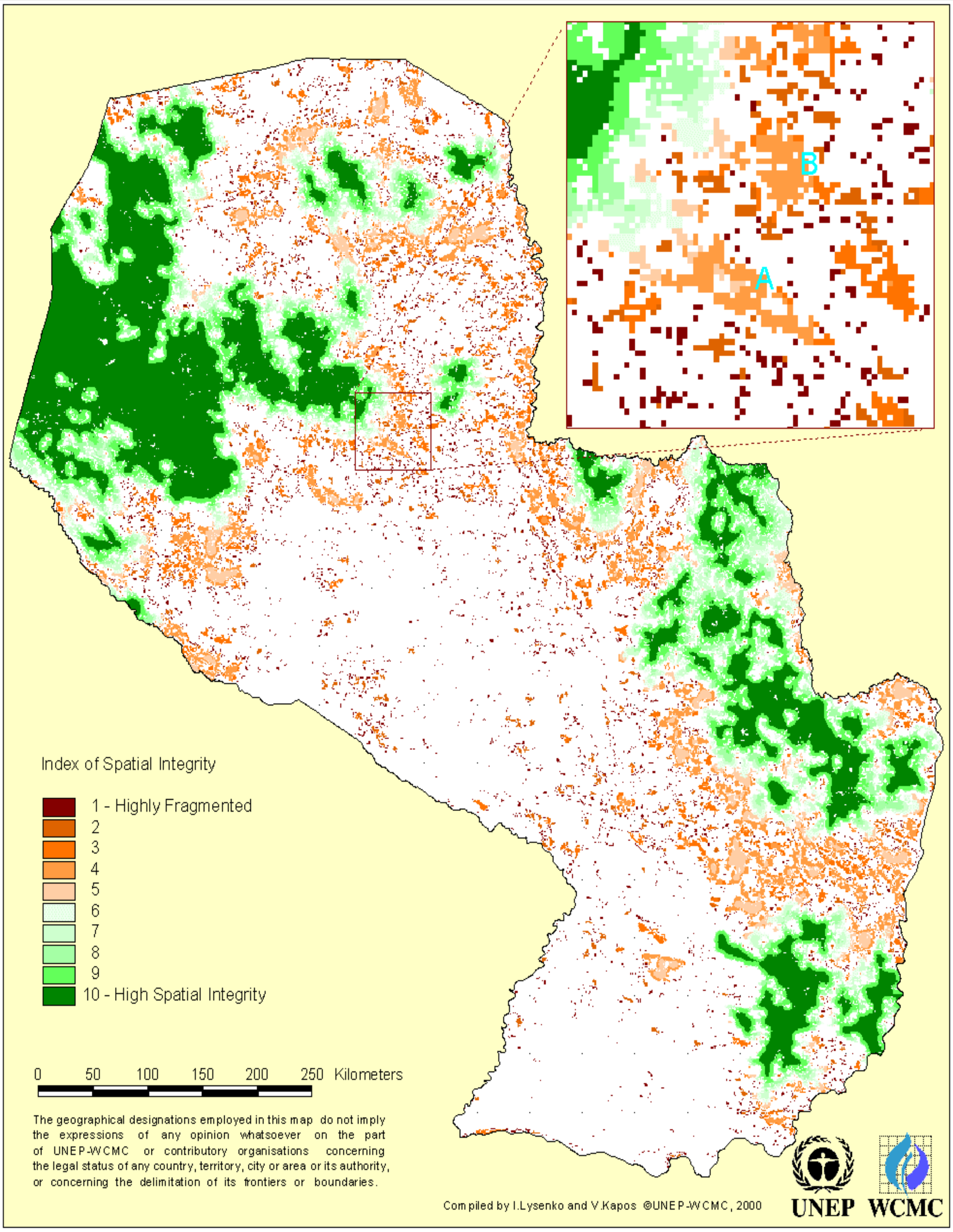

Figure 9. The forest cover of Paraguay displayed by integrity classes derived from the index of Forest Spatial Integrity $\mathbf{F_F}$.

The index reflects the size, shape and connectivity of each forest patch. It varies smoothly without being affected by small connections (near the resolution of the source data) to large forest patches [A], or separations from them [B]

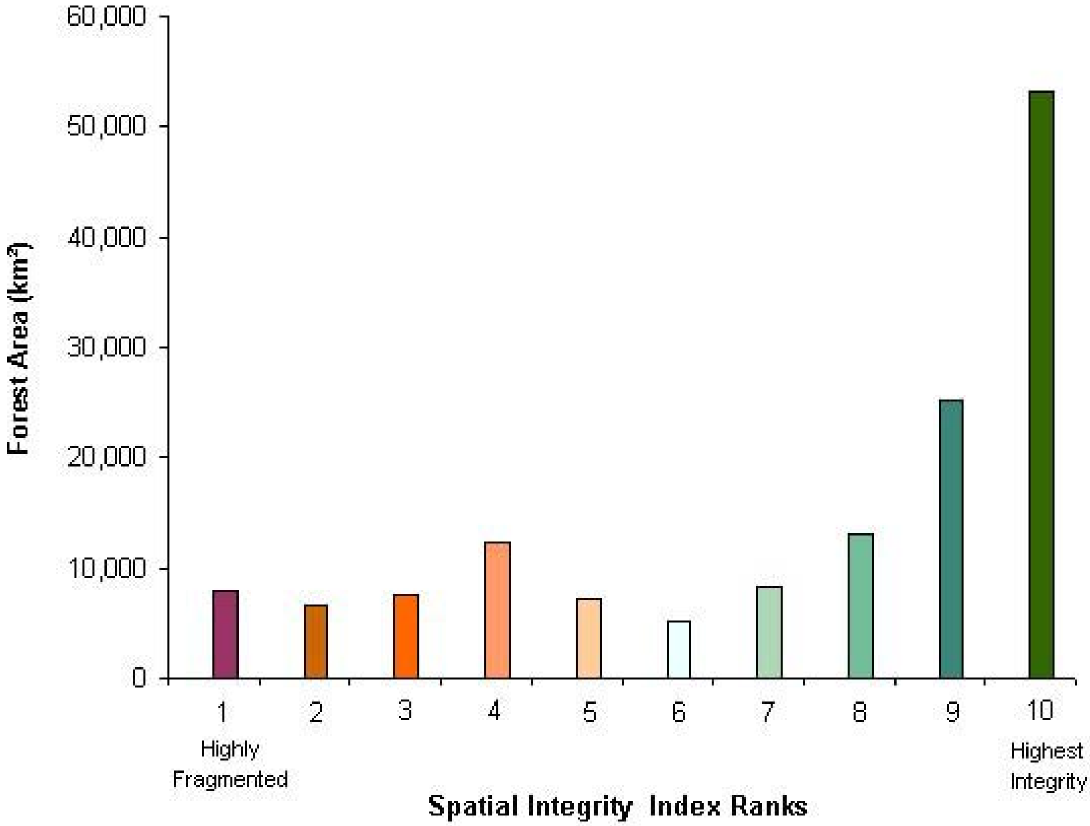

Figure 10. Statistical distribution of the forest area of Paraguay among different classes of spatial integrity as expressed by the Index of Forest Spatial Integrity, $\mathbf{F_F}$, which combines information on patch size, shape and isolation. Such a distribution can be used as a baseline for monitoring forest condition with respect to fragmentation. If deforestation reduces forest area uniformly, the totals will decrease uniformly across the distribution. The loss of small isolated forest remnants or irregular patches will be reflected in a reduction in the area in the lower integrity classes, while the loss of forest with high integrity can be detected from a loss of forest area with high integrity value. Distributions can be compared between assessment times by using non-parametric statistical tests.

Added clarity of both mapped and statistical presentation of data on forest spatial integrity may be obtained by delineating the three different forest density zones (Fig. 11) as derived from the simplified presentation of the Spatially Weighted Forest Density (Fig. 6). This presentation might help the non-expert user to visualise the spatial relations among forest patches in relation to their overall spatial integrity index and to anticipate the effects of changes in the landscape more vividly. It also makes it possible to view the statistical distribution among intermediate integrity classes in more detail (Fig. 12), as the scale can be expanded by excluding large areas of high and low integrity forest from the presentation.

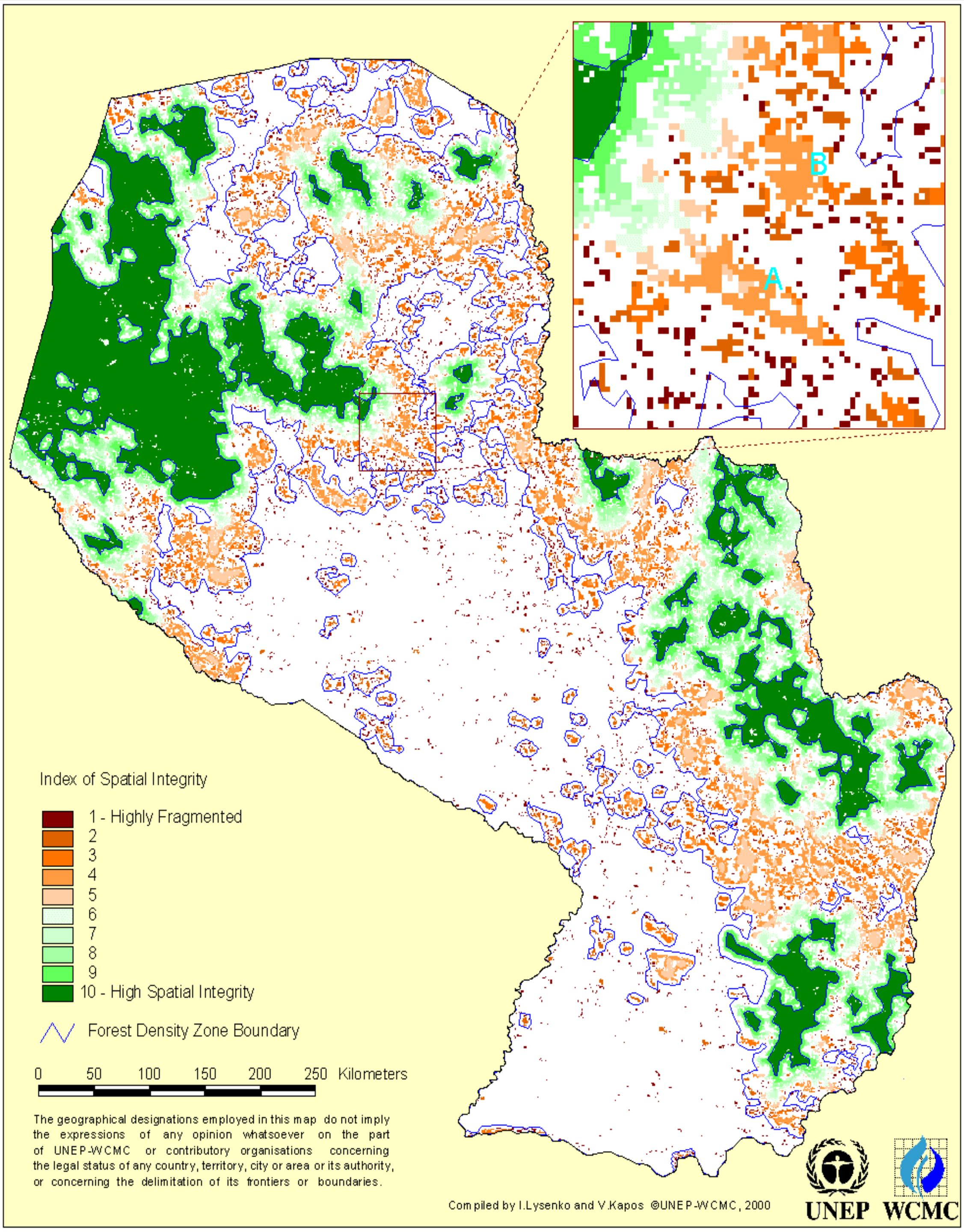

Figure 11. The forest cover of Paraguay displayed by integrity classes derived from the index of Forest Spatial Integrity $\mathbf{F}_F$, with added visual impact provided by delineation of the limits of the Forest Density Zones (Fig 6).

This approach might make it easier for the non-expert user to integrate statistical and spatial data and anticipate the potential effects of landscape changes that might result from policy or management decisions.

Figure 12a

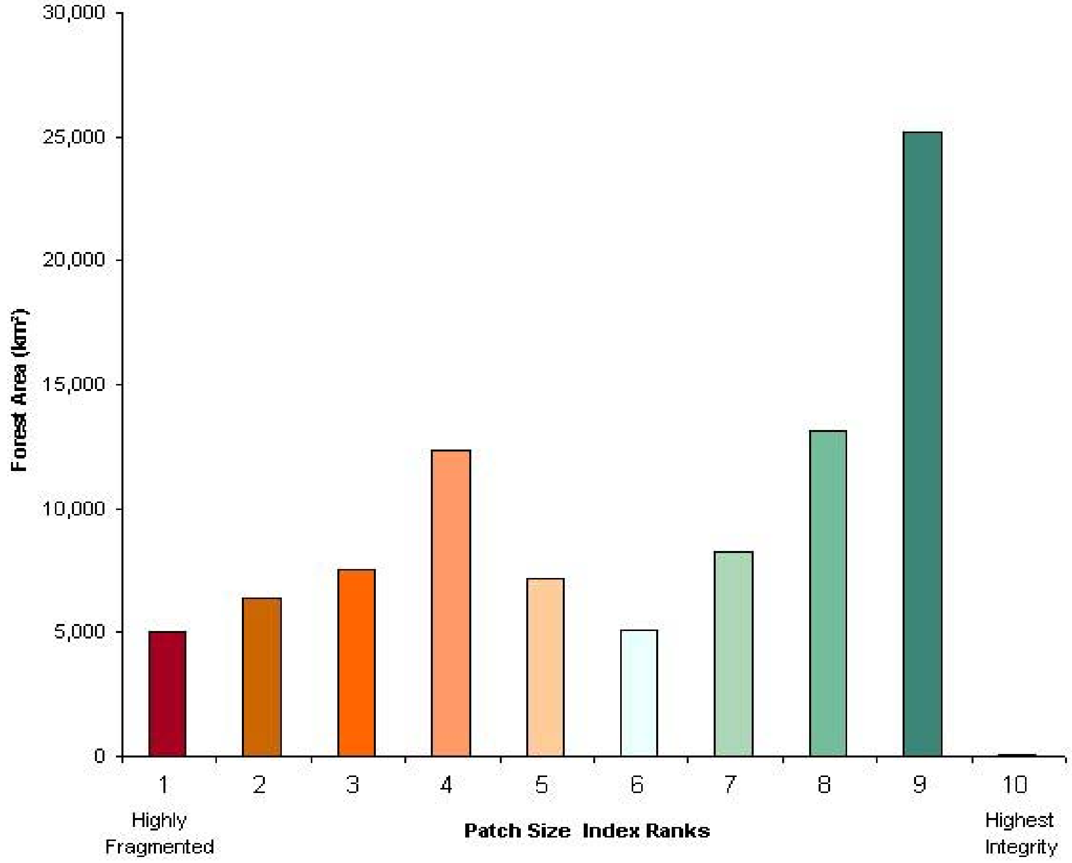

Figure 12b

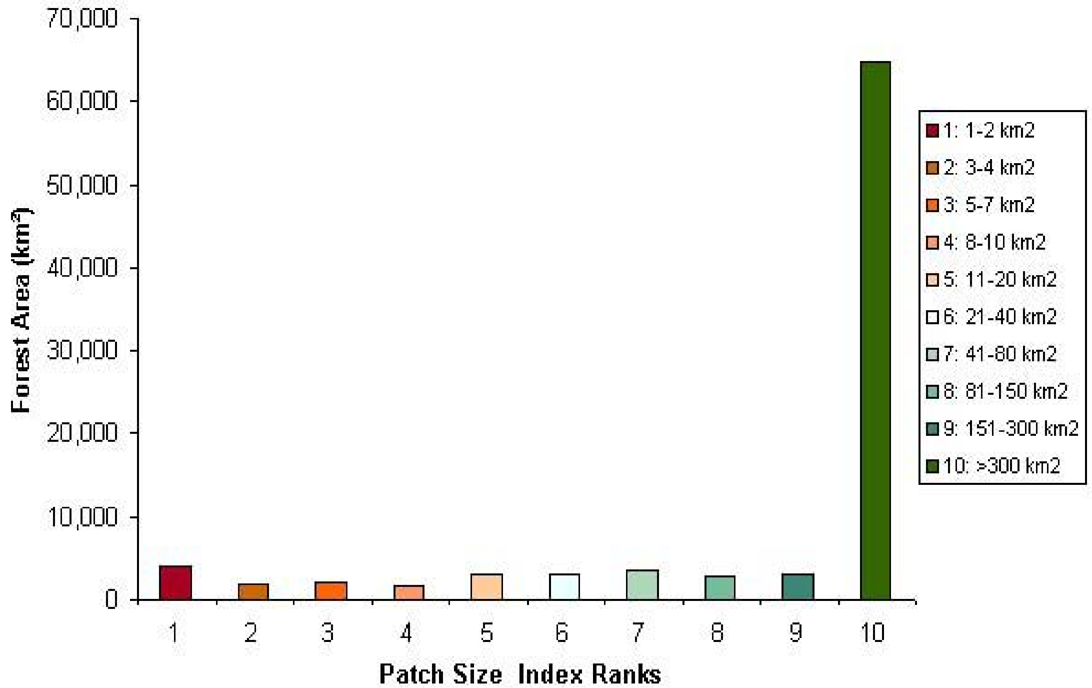

Figure 12c

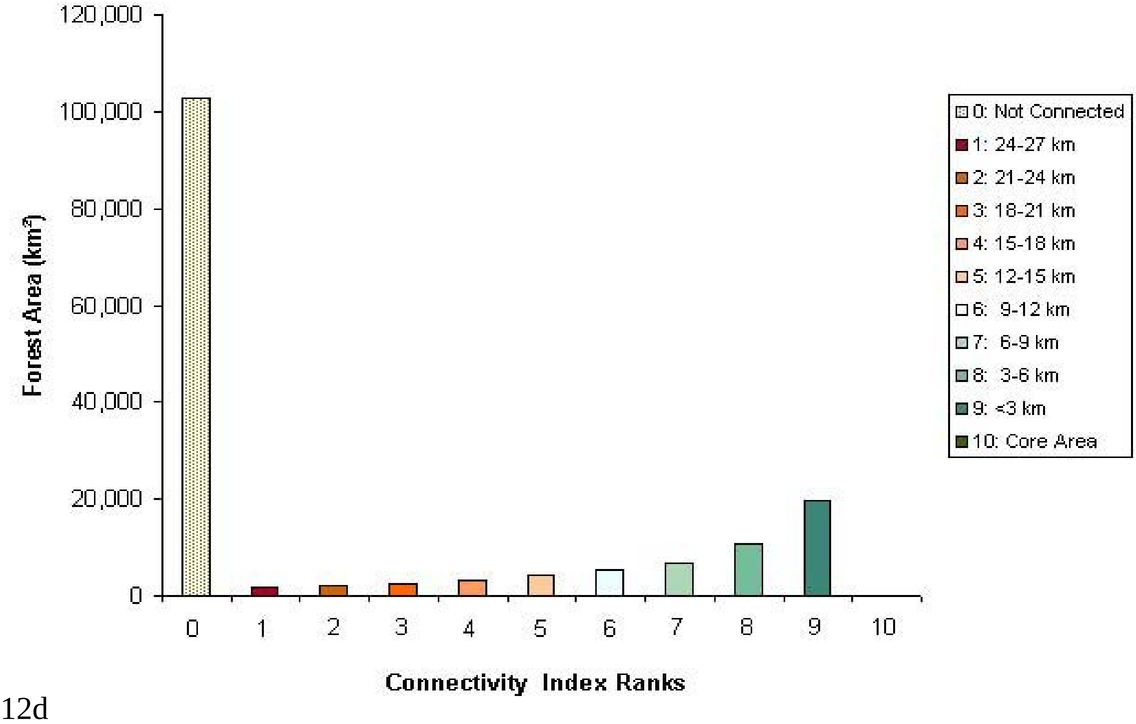

Figure 12d

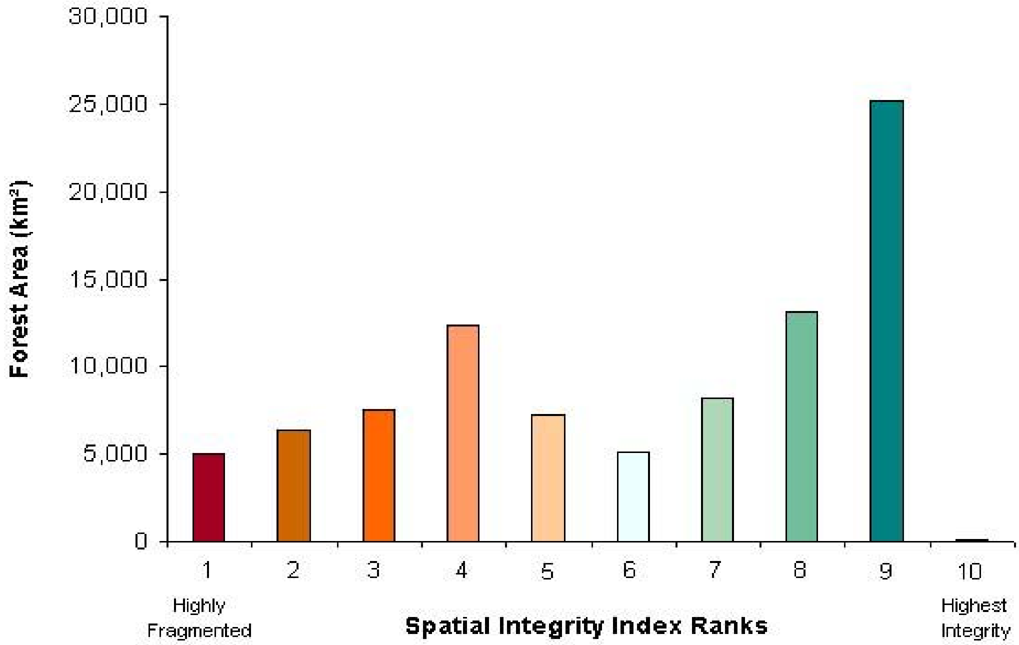

Figure 12. Statistical distribution of the forest area of Paraguay, according to the three component indices and the integrated Index of Forest Spatial Integrity. These graphs include only the forest falling within the zone of intermediate density (cf. Fig 6) and permit greater scrutiny of the distribution of forest among intermediate classes of each index and, potentially its change over time, than Figure 10. Forest areas of intermediate integrity are those most likely to be affected by changes in policy and management, and therefore such a form of presentation may be of use to inform decision-makers.

Another application, to data from the Ukraine (Figs 13 and 14), shows that the analysis produces visually consistent results and that it is feasible to apply this approach widely. It shows clearly that the Ukraine has smaller amounts of core and high integrity forest, but retains some areas of intermediate integrity that can be identified and prioritised in policies to promote forest conservation and sustainable use. This analytical approach could be implemented in a global assessment of forest fragmentation to provide a baseline for monitoring change in forest cover and its integrity and provide insight into the changing capacity of the world's forests to retain their biodiversity.

Spatial integrity is not in itself a sufficient measure of forest capacity to maintain biodiversity. Of the other influences that are important, a key factor is the influence of human population and activity on remaining forest areas. This is discussed in the following section.

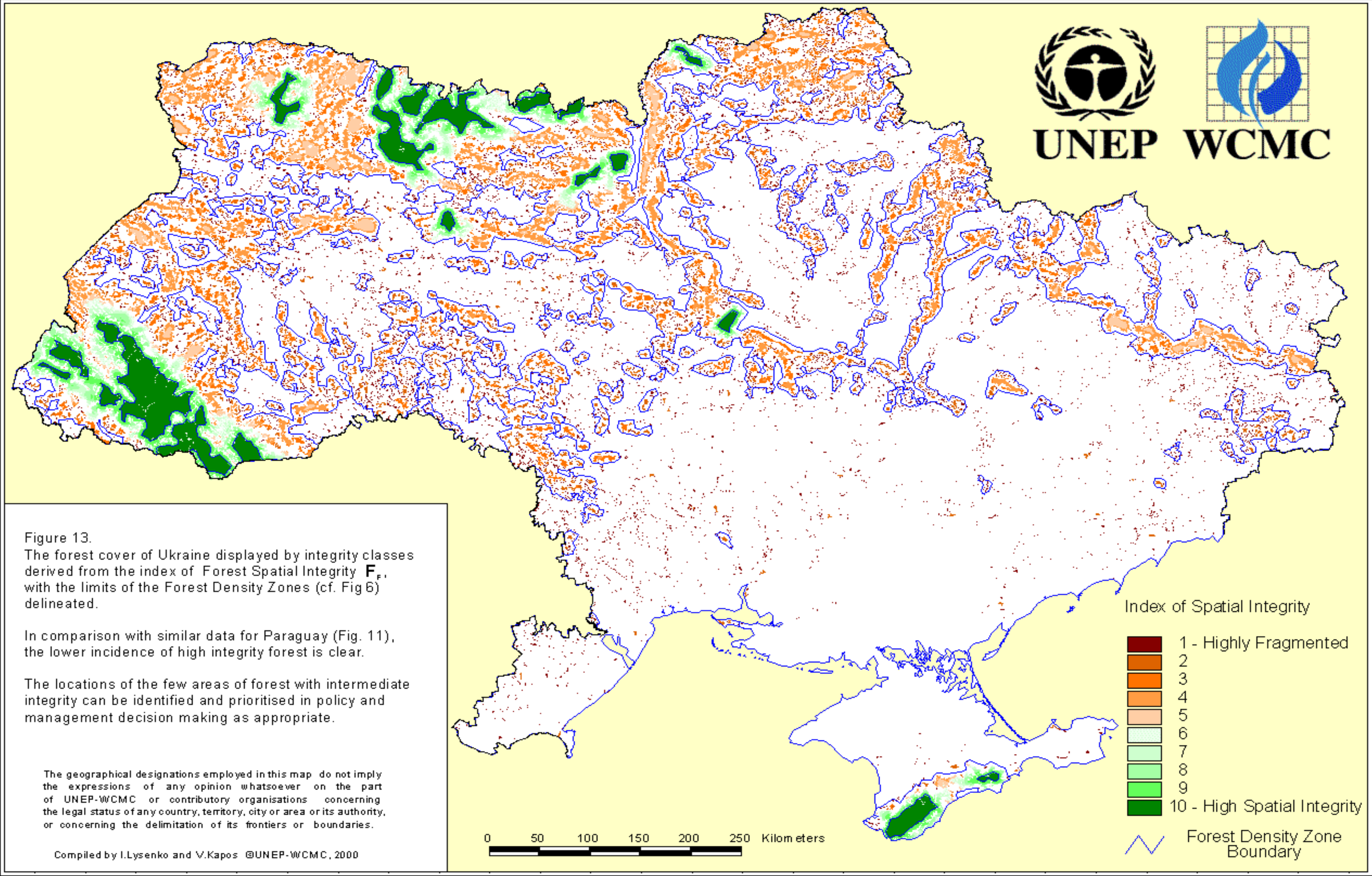

Figure 13.
The forest cover of Ukraine displayed by integrity classes derived from the index of Forest Spatial Integrity $\mathbf{F}_F$, with the limits of the Forest Density Zones (cf. Fig 6) delineated.

In comparison with similar data for Paraguay (Fig. 11), the lower incidence of high integrity forest is clear.

The locations of the few areas of forest with intermediate integrity can be identified and prioritised in policy and management decision making as appropriate.

Compiled by I.Lysenko and V.Kapos

Figure 14a

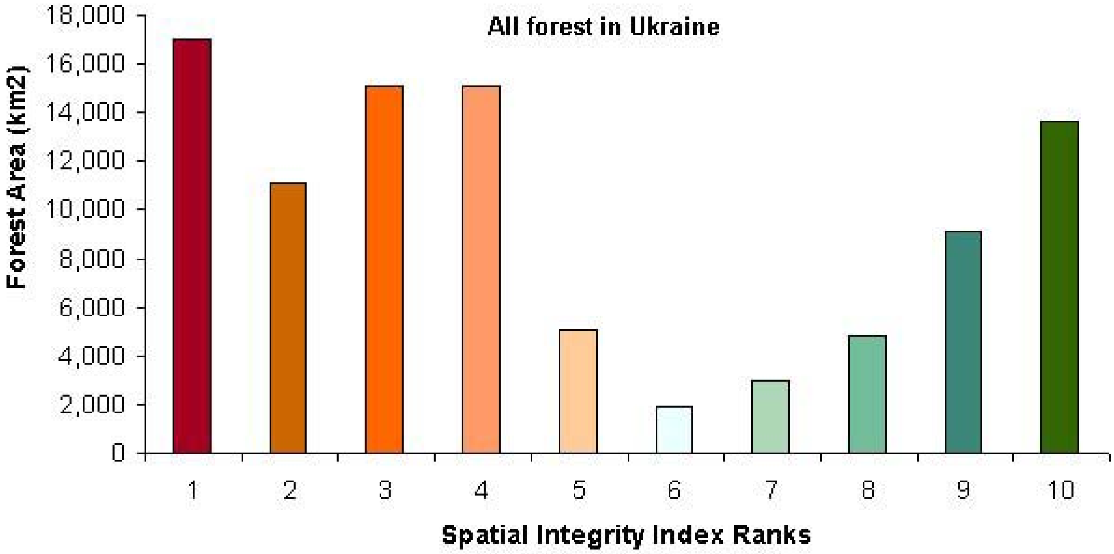

Figure 14b

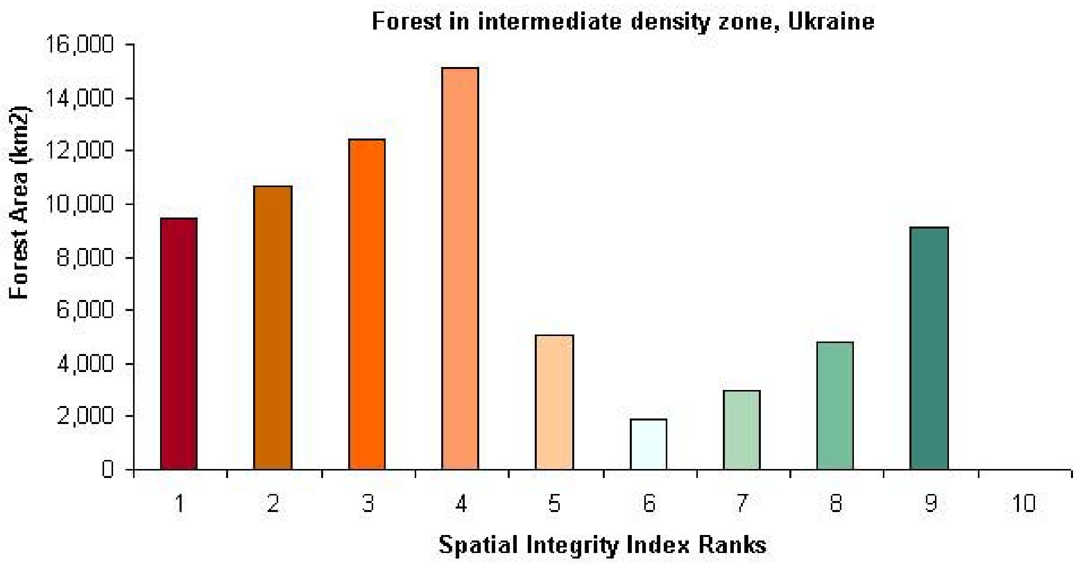

Figure 14. Statistical distribution of the forest area of Ukraine among different classes of spatial integrity as expressed by the Index of Forest Spatial Integrity, $\mathbf{F_F}$, which combines information on patch size, shape and isolation. Such a distribution can be used as a baseline for monitoring forest condition with respect to fragmentation or for comparison with data from other locations (e.g. Fig 11). In (a) the distribution is shown for all the forest area of the Ukraine, while in (b) only forest falling within the zone of intermediate density (see Fig. 13) is included, allowing more detailed scrutiny of the distribution among intermediate integrity classes.

IV A Focus on Human Activity

To this point discussion has focussed on methods for measuring change in the forest itself, insofar as this can be detected from coarse resolution data. An alternative strategy is to switch focus from ecological change to describing the factors that ***drive*** ecological response; in this case human activity itself (Lesslie 1997).

Shifting emphasis in this way means that generalisations can be made about the level and extent of human intervention in ecosystems and, in turn, the exposure or vulnerability of these ecosystems (Lesslie 1997). It also alleviates difficulties associated with identifying and measuring key ecosystem phenomena, facilitates an explicit treatment of scale issues, improves prospects of acquiring suitable data, and removes the necessity to ascertain whether particular ecological outcomes are attributable to human activity or natural processes.

Spatial Pattern

The analysis of spatial pattern in human activity in the landscape has a strong tradition in human geography, particularly from the 1930s to the 1970s, from which several key principles have emerged. Firstly, the spatial distribution of human activity reflects an ordered adjustment to distance. Of particular relevance is the notion of the attenuation of pattern or process with distance, as expressed in Tobler's 'first law of geography' which states that everything is related to everything else, but near things are more related than distant things (Tobler 1970). Secondly, human activity is generally located to minimise the 'frictional' effects of distance (Losch 1954). A related principle is the notion of accessibility or functional centrality. Finally, human activity agglomerates in settlements.

Generalisations concerning the spatial configuration of habitat (Noss & Cooperrider 1994) are encapsulated in the broader generalisation that the integrity of habitat is usually associated with spatial isolation from human activity. This broader principle also relates to more traditional geographical perspectives on the pattern of human activity in the landscape and the attenuation of pattern and process with distance. However, a reliance on spatial pattern to explain human interaction with ecosystems does have limitations. It involves, for instance, the necessary assumption that there is a direct relationship between the spatial location of human activity and its ecological effects. This precludes any satisfactory accounting for processes (e.g. hydrologic and atmospheric) where spatially distributed effects are non-linear or highly complex (e.g. multi-scale processes).

Measuring Isolation from Human Activity at Landscape Scales: Some Previous Experience

Spatially explicit indicators of ecological integrity, which are consistent with a strategy based on measuring isolation from human activity in the landscape, should have the following characteristics:

- The index should be quantitative and the methodology should have the capacity to measure variation in exposure to human activity across the landscape.
- Index values should derive directly from primary attribute data in a systematic and repeatable manner, and they should reflect the scale characteristics of primary data inputs.
- The methodology should be transparent and as simple as possible. Notions of ecological integrity and naturalness (or isolation from human activity) are complex and difficult to deal with in a precise way. There is, therefore, little advantage gained from pursuing complex modelling techniques when these are likely to have deficiencies of a similar order of magnitude to simpler procedures. Complex modelling also has the disadvantage that it can become difficult to understand and interpret. Any modelling procedure in this area will have contentious aspects and it is far better for these to be explicit and well understood. Complex modelling also generally requires primary attribute data of accuracy and precision that is typically not available at global or regional scales. Global consistency and comparability is improved if the modelling procedure does not make demands for sophisticated primary attribute data.
- The methodology should be amenable to elaboration in a staged and systematic manner. This is essential to enable the model to benefit from new or improved attribute data and knowledge. A capacity for elaboration may also be useful in local or regional situations where its possible to take advantage of additional or enhanced local attribute data, or where particular local factors have a known and significant impact on ecological integrity.

Large Natural (Roadless) Areas

All other factors being equal, the core of a large natural area will be less exposed to (more isolated from) human activity than the core of a small natural area. The exposure of a natural area to human activity is therefore fundamentally related to its size.

Current interest in the size of natural areas, as a measure of relative isolation from human activity, is complemented by its long history in nature conservation science. Since the 19th century, for instance, it has been recognised that there is a link between the human-induced break-up of large natural areas and the extinction of species (De Candolle 1874). The importance of size in nature conservation science has, however, been promoted most strongly by island biogeography theory (MacArthur & Wilson 1967) and the species-area relationships of Preston (1962), which hold that larger areas typically support a greater diversity of habitats and contain more species and larger populations of individual species than smaller areas. Size is today similarly recognised as important in process-functional terms. Larger natural areas have the capacity to better accommodate change in larger-scale processes (eg shifts in climate patterns) and disturbance events (eg fire) (Forman 1996).

The systematic evaluation of natural areas for conservation, based on size, is first evident in the US early in the 20th century with the emergence of interest in wilderness and roadless areas. During the 1930s the US Forest Service prepared an inventory of available wilderness areas within National Forest, rating undeveloped roadless areas on the basis of size. The US Wilderness Act of 1964 set up a framework for wilderness identification and protection in the US that is still in place today. It requires relevant Federal land-holding agencies to identify and assess roadless areas.

Interest in the identification of large natural areas that have limited exposure to modern technological society is not restricted to the US. It has been a mainstream aspect of nature conservation assessment in other 'frontier' regions of the world where there is a relatively clear distinction between the presence and impact of human activity and natural ecological patterns and processes. This is especially the case in Australia, Canada and New Zealand. Notably, the concept has also been pursued in parts of the world where this distinction is not so clear and where there is continuing indigenous habitation. This includes countries such as Finland (Kajala & Watson 1997), South Africa (Elliot 1996), and Italy (Zunino 1995). Identification and assessment methodologies typically involve criteria that specify minimum size and shape characteristics and require a 'primary' vegetation cover, no urban, agricultural or other commercial land use, minimal constructed access and no permanent settlement. Detailed specifications vary on a regional basis and from study to study.

Several assessments of this type have been conducted at the global level. The first global 'reconnaissance-level' assessment of large areas with minimal impact and proximity to modern technological society was completed by McCloskey and Spalding (1989). Using 1:1,000,000 scale Jet and Operational Navigation Charts as a database, that study identified areas of at least 400,000 ha with no mapped human structures or roads. The global distribution of these areas is (Figure 15) is heavily dominated by arctic and desert regions and includes little area in forested zones.

A second global-level evaluation of human activity (Hannah *et al.,.* 1994) mapped relatively undisturbed (not simply roadless) natural areas, and reduced the minimum size of mapped 'undisturbed' areas to 100,000 ha. It defined undisturbed and two classes of non-natural areas as follows:

- ***undisturbed*** - a record of primary vegetation and a very low population density (<10 persons per km^2 density or <1 per km^2 density in arid/semi-arid and tundra communities)

- ***partially disturbed*** - a record of shifting or extensive agriculture, or other record of human disturbance; and

- ***human dominated*** - a record of permanent agriculture or urban settlement, or where primary/potential vegetation is removed

Data were derived from a variety of information sources. Overall, the world was found to have around half of its total surface, but only 27% of its habitable surface undisturbed by man. A number of important forest areas, especially in the Indomalayan biogeographic realm, had no undisturbed area remaining and very little partially disturbed territory.

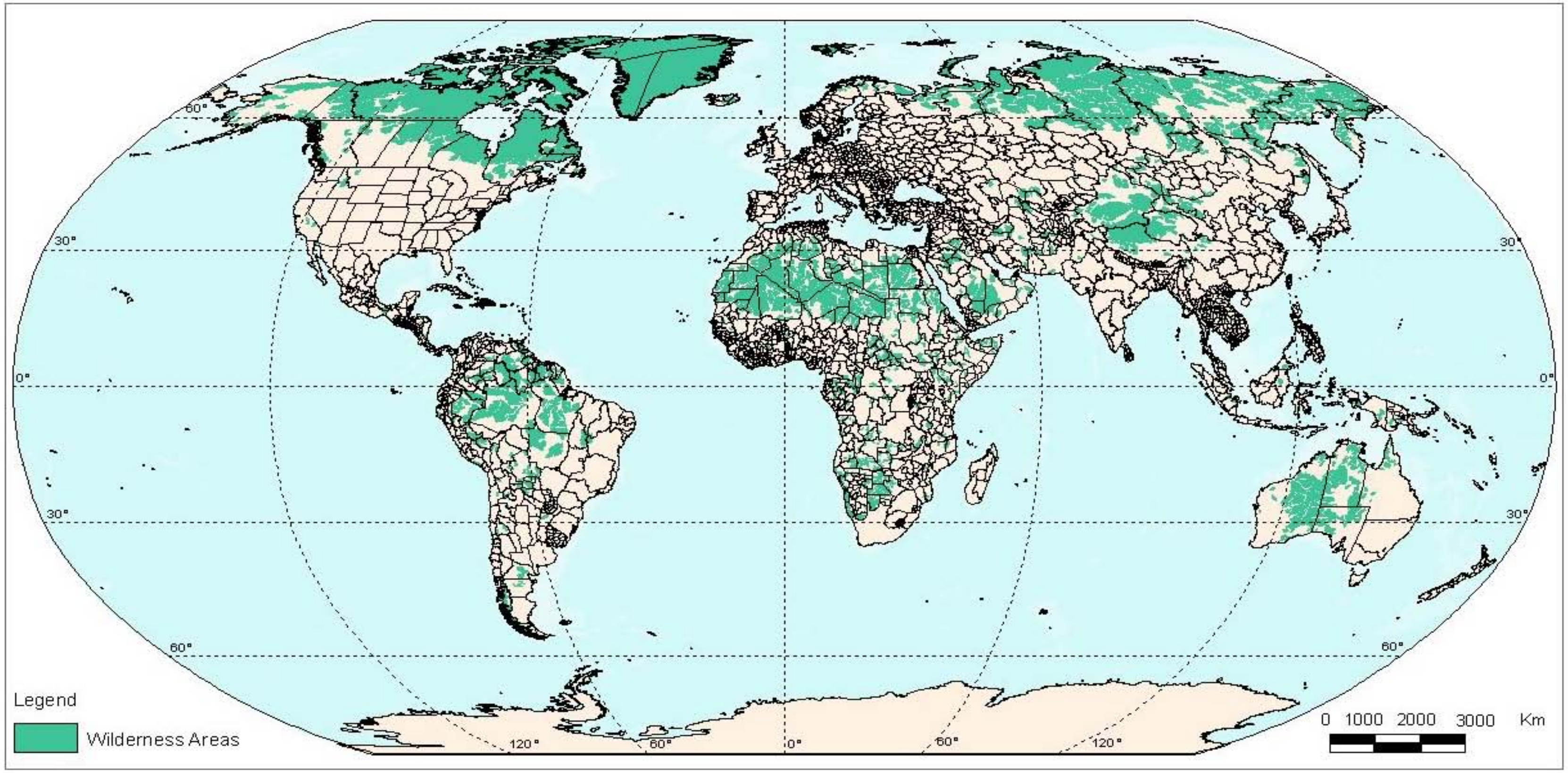

Figure 15: World wilderness areas (i.e. at least 400,000 ha with no mapped human structures or roads) as identified by McCloskey and Spalding (1989) using the Jet and Operational Navigation Charts. Data from the Sierra Club and World Bank, integrated and supplied by UNEP-GRID, Geneva. (http://www.grid.unep.ch/datasets)

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The geographical designations employed in this map do not imply the expressions of any opinion whatsoever on the part of UNEP-WCMC or contributory organisations concerning the legal status of any country, territory, city or area or its authority, or concerning the delimitation of its frontiers or boundaries.

An evaluation of the status of the world's 'frontier forests' represents a third instance of a global-scale study involving the identification of relatively large natural areas (Bryant *et al.,*. 1997) (Figure 16). This assessment focuses solely on forest environments, distinguishing three classes of threat:

- ***frontier forests under low or no threat*** - large intact forest ecosystems that are relatively undisturbed and large enough to maintain their biodiversity
- ***frontier forests under medium or high threat*** - on-going or planned human activity (eg logging, agricultural clearing, mining)
- ***non-frontier forests*** - secondary forest, plantations, degraded forest, and patches of primary forest.

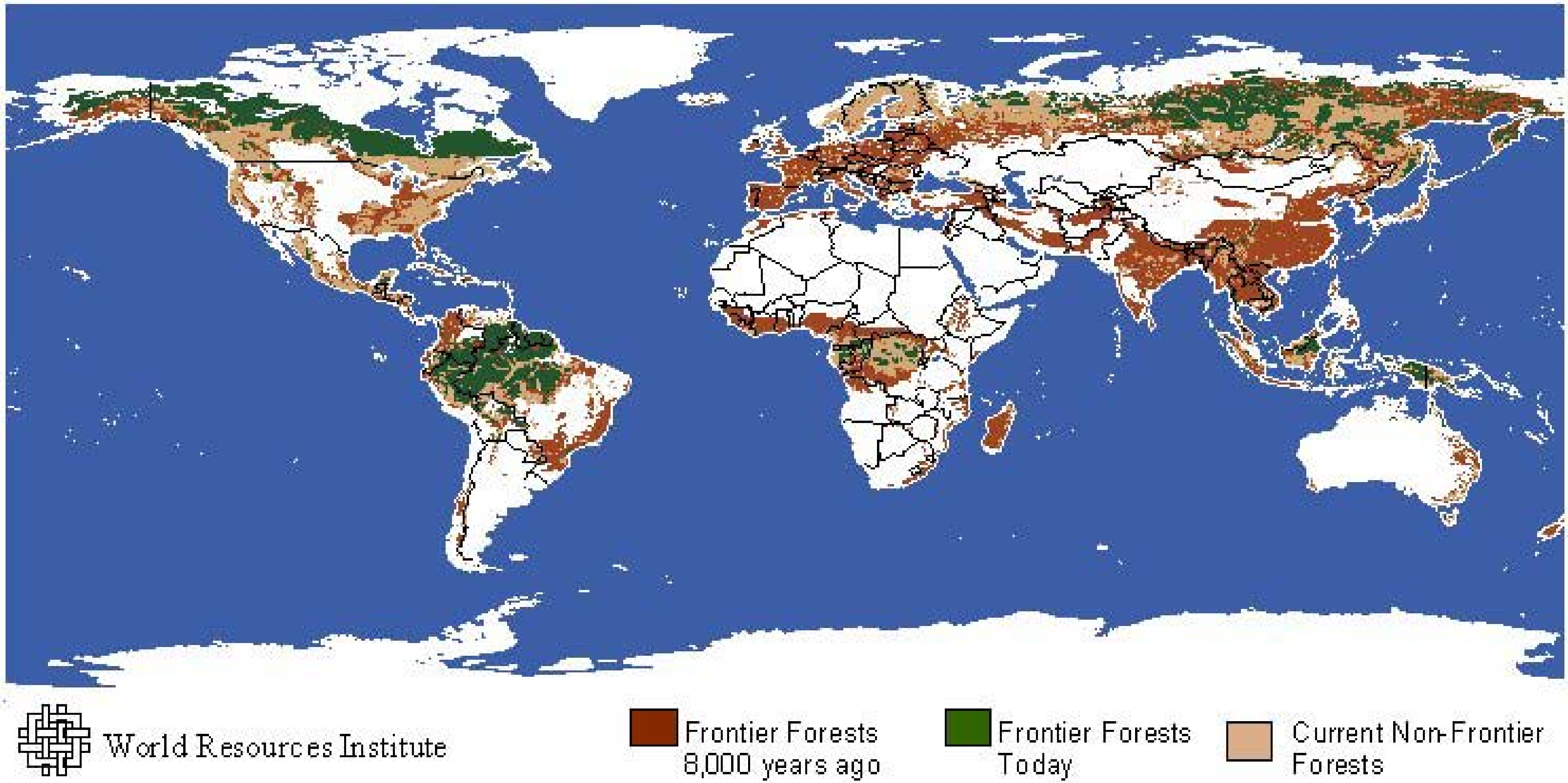
The threat classification in this mapping exercise was drawn largely from expert opinion. No specific size threshold was applied in identifying frontier forests, although it was required that areas be of sufficient size to maintain biodiversity and to absorb large-scale disturbances.

These three global-scale spatial studies represent simple and useful assessments of the naturalness or integrity of ecosystems. Each study primarily relies on the assumption that isolation from the impacts and influences of human activity is a reasonable indicator of ecological integrity, although some reference is made to certain biophysical conditions in the latter two instances. In each case the accuracy and precision of results is dependent upon the quality of available suitable data of global extent.

A greater concern, particularly in the latter two studies, is that the analyses are not explicitly scaled; nor are they systematic and repeatable. This means that the precision of the mapping and the accuracy of attribute class allocations are not transparent and direct expressions of the data and the analytical procedure. It also means that they can not form the basis for any kind of consistent assessment programme to monitor change through time.

Moreover, the selection of particular size thresholds to identify places that are isolated from the impacts and influences of human activity can be questioned. Exposure to human activity is not simply a matter of presence or absence, it is a matter of degree.

Figure 16 – Forest Frontiers Map



Measuring a Continuum: the Wilderness Index

Recognition of problems with the use of fixed size thresholds, qualitative terms and a lack of repeatability gave rise, in the early 1980s, to new quantitative indicator-based approaches to the identification of remote natural areas. For example, emphasis switched from the identification of wilderness areas based on qualitative thresholds to the 'continuum' concept of wilderness (Lesslie and Taylor 1985), involving the quantitative measurement of relative variation in remoteness from human activity across the landscape (Kirkpatrick & Haney 1980; Lesslie *et al.*, 1988). This type of approach underpinned the Australian government's National Wilderness Inventory (Lesslie & Maslen 1995) and the production of similar remote and natural lands databases elsewhere (e.g. Husby 1995).

The Australian wilderness study places emphasis on measuring the extent to which points in the landscape are remote from, and undisturbed by, the influence of modern technological society. It does so by quantitatively measuring variation in remoteness and naturalness across the landscape using four indicators:

- ***remoteness from settlement*** (remoteness from places of permanent habitation);
- ***remoteness from access*** (remoteness from established access routes);
- ***apparent naturalness*** (the degree to which the landscape is free from the presence of permanent structures associated with modern technological society); and,
- ***biophysical naturalness*** (the degree to which the natural environment is free from biophysical disturbance caused by the influence of modern technological society).

The two remoteness indices and the apparent naturalness index are based on a measurement of Euclidean distance between each point in the landscape and ordered classes of settlement and infrastructure. Variations in exposure to different types of settlement and infrastructure features are accommodated through a weighting and distance-decay regime whereby more prominent feature types (such as highways or commercial centres) are accorded greater influence than less prominent types (such as vehicle tracks or residences). In this way, distance-based measures represent spatial pattern that is specific to the location of individual landscape features, allowing for the attenuation of levels of technological activity according to the distance from the feature and its prominence or likely influence. The use of distance measures in this context is not based on any empirical relationship between distance and the flow of resources associated with types of technological activity.

The biophysical naturalness index is rated according to the intensity of land use in areas where the primary vegetation structure is essentially intact. Land use, in this context, refers to activity that is not confined to defined physical structures (settlement and infrastructure) and includes a variety of forms of spatially distributed resource procurement activity, such as timber production and livestock grazing. Land use intensity is rated on the basis of historical records or of land use likelihood modelling.

The distribution of wilderness quality across Australia, or Australian Wilderness Index (AWI) was obtained by linear un-weighted combination of the four component measures as illustrated in Figure 17.

This type of approach has a number of advantages over conventional mapping methods. Of particular relevance to the question of periodic assessments and monitoring is the fact the analysis is quantitative and repeatable. Estimates of isolation or exposure to human activity produced by the analysis are a direct expression of the data and the modelling that is applied. This means that the scale of the analysis can be explicitly matched to the accuracy and precision of data inputs. GIS-based application of the model, which effectively automates the analysis, also promotes flexibility so that new primary attribute data can readily be introduced and the analysis repeated or manipulated in a variety of ways.

Measures of Accessibility, Population Density and Resource Use

The Australian wilderness index measures environmental exposure or isolation from human activity in the landscape in terms of Euclidean distance from classes of settlement and infrastructure, along with a rating of land use intensity. No attempt is made to represent exposure or isolation in ways that are more exact or 'real'. The notion of exposure to, or isolation from human activity may be elaborated in two ways: 1) the refinement of the spatial, distance-based aspects, and 2) enhanced calibration of the intensity of human activity, taking account of data relating to appropriation and use of resources.

One obvious way in which the spatial, distance-based component of isolation may be enhanced is through use of more refined measures of accessibility. The accessibility of places in the landscape is not simply a function of Euclidean distance from access points and the quality of that access. Accessibility is also dependent, for example, on terrain. For instance, a forest that is located in rugged terrain at a given distance from a road is generally less accessible to timber harvesting than a forest that is located at the same distance from a road in flat or undulating terrain. The improving quality of digital elevation modelling and the increasing availability of elevation data sets means that it is now practicable to introduce terrain factors into accessibility modelling at region and global scale. A global digital elevation data set (DEM) modelled from satellite imagery is, now available at a grid resolution of approximately 1 km^2 (USGS EROS Data Center 1996). The accessibility of forest environments has recently been modelled at regional scale using slope measures derived from elevation data at that scale (Lorenzini 1998).

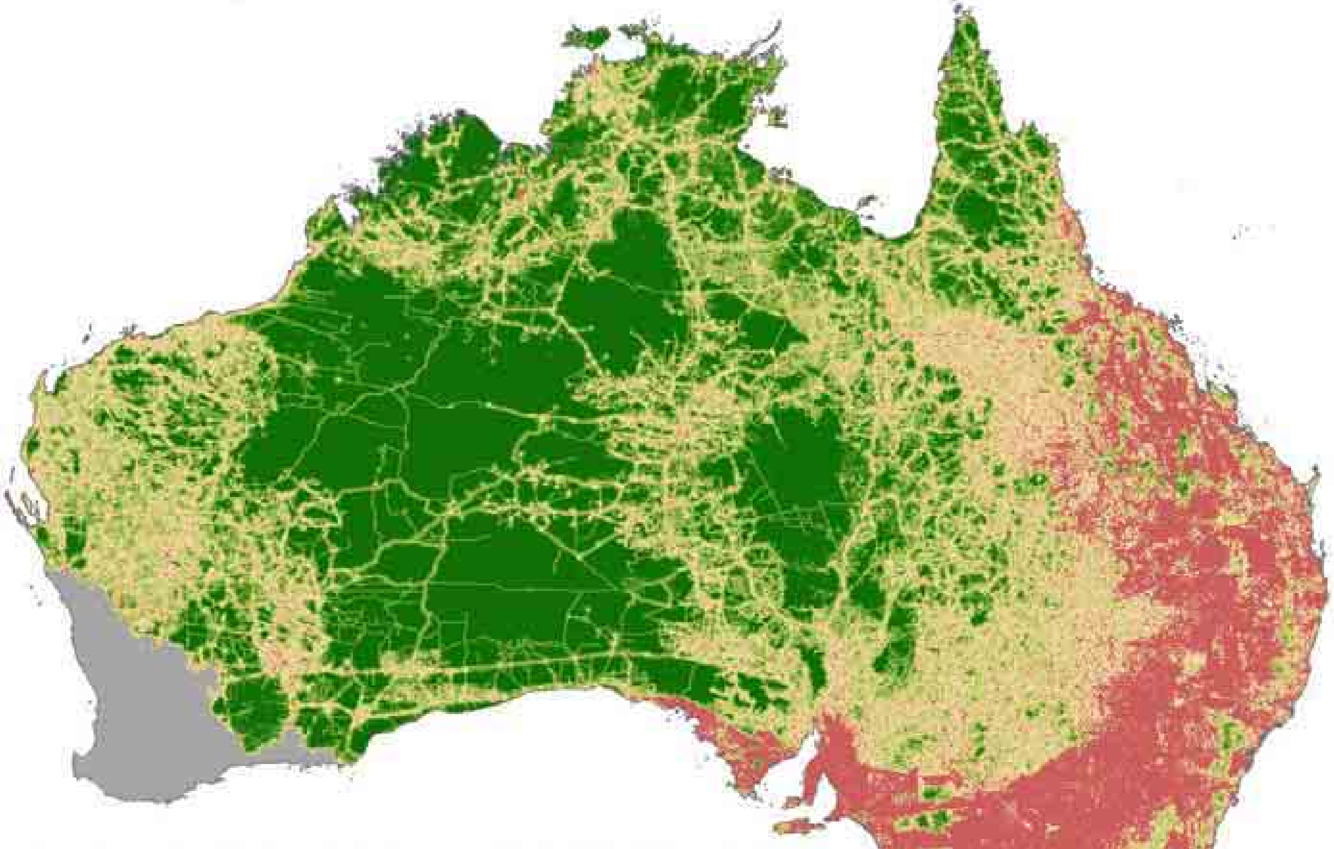


Figure 17. The land surface of Australia classified according to the Australian Wilderness Index (AWI, Lesslie & Maslen 1995). The AWI was calculated on a 500-m grid by combining each cell's euclidean distances from settlements, access and other infrastructure features with a biophysical naturalness score derived from land use data. The resulting continuum of wilderness values reflects the likely degree of human influence at each point on the continent and is heavily influenced by the information on biophysical naturalness.

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Enhancements of this kind should, however, be treated with caution. There is no doubt that terrain-related factors have significant impact on accessibility in the real world, but terrain attributes are highly sensitive to DEM accuracy and precision, and estimations should be treated very cautiously. Typically, a DEM grid resolution in the order of 20 m is required if the tilt components of topographic variation are to be faithfully reflected in model output (Moore *et al.*,. 1993). It is therefore doubtful that terrain data at the scale presently available have the capacity to add meaningfully to accessibility analyses of the kind under discussion. However, the prospects for this type of analysis will certainly improve over time.

The second area of model elaboration involves calibration of the intensity of human activity. One simple way this may be developed is by factoring in statistical data on population and spatially relating this to settlement and infrastructure patterns. Indeed, patterns of population density have already been derived in this way on a global basis and show promise in this regard (Tobler *et al.*,. 1995).

The calibration of the intensity of human activity could potentially be further refined by combining population data with information about resource use. The intensity of demand pressure on forest ecosystems may, for example, be calibrated by inclusion of demand estimates for forest resources. A demand surface for fuelwood could be developed by using statistical data on fuelwood consumption, and distributing this spatially around urban and rural settlements on the basis of population density, forest cover, accessibility and known patterns of fuel wood consumption. A similar surface representing the pressures caused by industrial demand for timber resources could be developed and spatially distributed through settlement, infrastructure and land use components.

Developing Spatial Indicators of Naturalness for Use at Global and Regional Scales

Discussion to this point reveals some key principles that guide the development of spatial indicators for assessing the naturalness of forests.

- Indicators based on ecological response to human perturbation can only provide limited, and possibly contradictory, answers to questions concerning ecological change at the ecosystem level.
- Indicators of naturalness that focus on human activity - the driver of human-induced ecosystem change - are highly promising in generic, landscape-scale applications. Such indicators rely on the assumption that the greater the exposure to human activity, the greater the probability of human interaction and intervention in ecosystems.
- Spatially explicit measures of relative environmental isolation from human activity can provide a generic foundation for describing and measuring the potential for human intervention in ecosystems.
- Methods for measuring relative environmental isolation should be quantitative, repeatable, transparent, appropriate to the input data, simple to interpret and amenable to elaboration.

Key Attributes

Human activity in the landscape can be described in terms of spatial patterns of land use and land occupation. Categories of landscape modification can be represented unambiguously in terms of (1) settlement, (2) infrastructure, and (3) land use.

1. **Settlement** can be defined as permanent human occupation. It is the focal point for human activity in the landscape where resources are transformed and used. Settlements range in scale from a single point of permanent occupation, such as a house, through to conurbations which may extend over thousands of square kilometres.

2. **Infrastructure** is the built fabric around which human activity concentrates. Infrastructure provides the physical means for accessing, distributing and transforming resources. Infrastructure includes all built structures, including those associated access and settlement.

3. **Land use** includes any resource procurement or transformation activity that can be spatially delimited on the land surface.

The representation of human activity using primary data sets comprising settlement and infrastructure features and land use has a number of advantages.

- These features represent fundamental elements through which the pattern of human activity in the landscape can be measured and described at both small and large scale, encapsulating complex resources procurement and transformation processes.
- These features are unambiguous landscape phenomena, which are amenable to classification and measurement. This provides for some control over accuracy and precision in spatial (and temporal) representation. It also facilitates analyses that use spatial information technologies.
- These features provide flexibility in relating human activity in the landscape to ecological effects, allowing for either aggregated or disaggregated analyses.

Using settlement, infrastructure and land use features for representing human activity does, however, have the important limitation of excluding resource manipulation techniques that rely on naturally occurring physical or biotic phenomena, such as fire or specific plants and animals, commonly associated with indigenous societies. In modern societies at least, there may be some association between the distribution of these factors and discernible patterns of settlement, infrastructure and land use.

An Effective Spatial Model: Basic Requirements

Spatial analytical technologies such as GIS provide powerful tools for modelling patterns of human activity in the landscape. Established theory connected with the spatial structure of settlement and urban structure provides also some useful principles to guide development of a modelling framework for representing spatial isolation from technological activity in the landscape.

A set of spatial indicators that represent a gradient of exposure to technological activity can be developed using distance-decay functions to rate locations across the landscape on the basis of their relative **exposure to settlement, exposure to infrastructure,** and **land use intensity**. Relatively high levels of exposure to technological activity are encountered in places that are close to urban areas. Relatively low exposure is associated with places that are isolated from the settlement, infrastructure and land use practices associated with modern technological society (Lesslie 1997).

The procedures devised for measuring the Australian wilderness index (Lesslie *et al.*,. 1988, Lesslie & Maslen 1995) offer a simple and coherent starting-point for measuring exposure to technological activity at the global scale (Figure 18). In the first instance, data on settlement, access and infrastructure features have been drawn from Digital Chart of the World, and the analysis conducted using a grid resolution of 2.,500 m. No land cover or landuse information was available for use in this analysis, so land use intensity is not included in this pilot analysis. However, in future an appropriate surface could be derived on the basis of elementary land use and land cover information.

This representation of naturalness can then be applied to forest area to provide a characterisation of forest naturalness, which can be displayed in both mapped and statistical forms.

Validation

Any attempt to link ecological integrity or naturalness with measures of exposure to human activity raises the issue of validation. It has already been stated that an emphasis on human activity alone means that specific ecological impacts cannot be inferred. However, some form of validation is critical in order to establish the extent to which there is agreement between the model and the real world. The validation process consists of showing that a model accords with facts as known, with what is accepted as true or reasonable, or is justifiable and appropriate for a stated set of purposes (Caswell 1976).

One attempt at validating the AWI approach, which was carried out at WCMC (Kapos 1997), focused on forest reserves in Uganda and Sri Lanka. The study compared a number of measures, including average wilderness scores and surrounding human population density, with expert on-the-ground evaluations of the relative condition or naturalness of each reserve. The AWI averages were the measure most closely correlated with the expert evaluations, and were more effective than, for example, human population density in predicting forest condition. This suggests that the approach may well be appropriate as an indicator of forest naturalness at broader scales. Additional validation exercises would be useful.

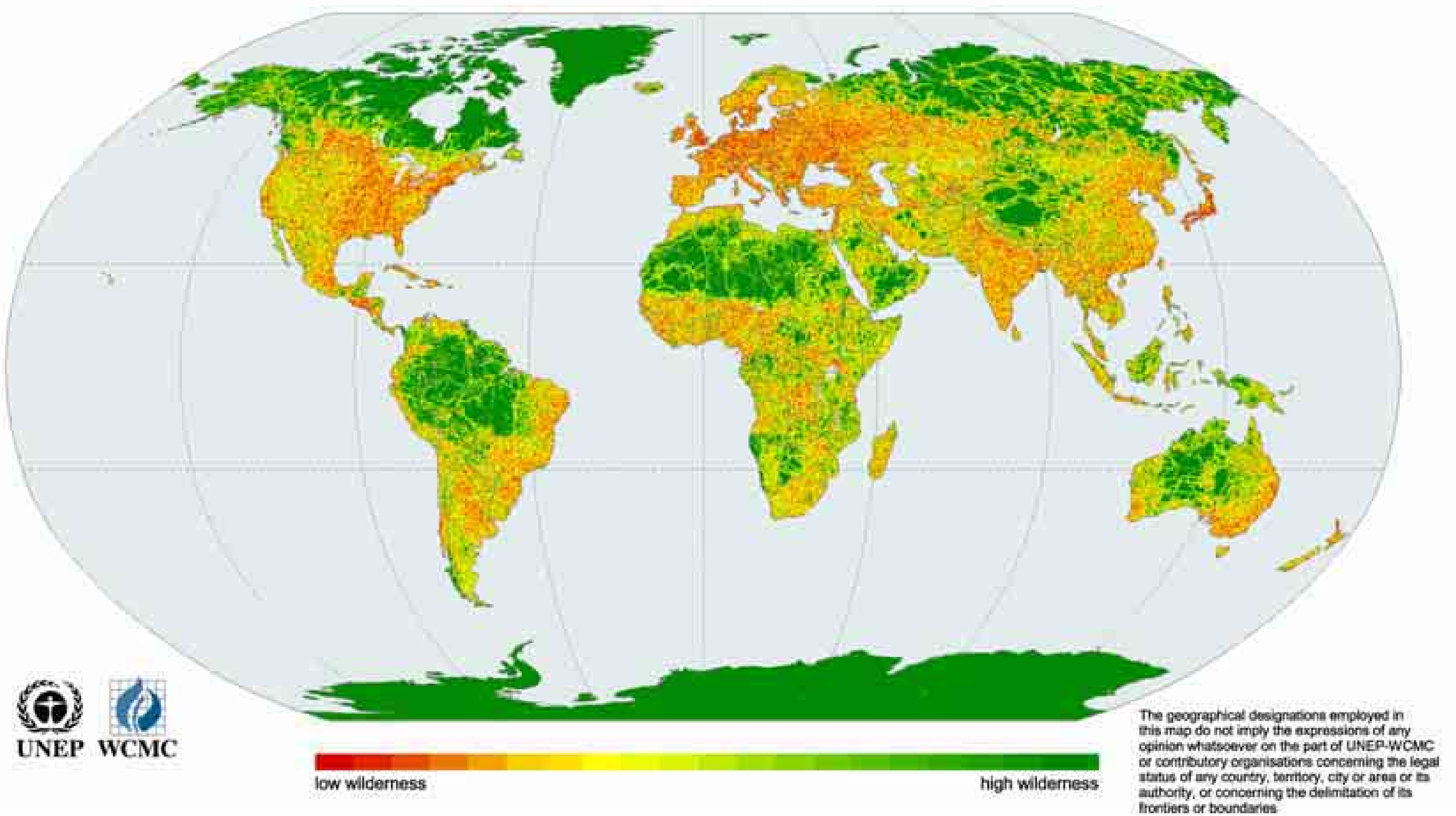

Figure 18: Global distribution of wilderness or ecosystem naturalness derived using the Australian Wilderness Index approach (Lesslie & Maslen 1995). Data on settlement, access and infrastructure features have been drawn from Digital Chart of the World, and the analysis was conducted using a grid resolution of 2,500 m. No land cover or landuse information was included in this pilot analysis. Such a surface can be overlaid with a map of the global distribution of forest cover to generate a global map of forest naturalness and associated statistical data.

Primary Attribute Data Availability and Quality

Primary attribute data availability is a key issue in determining appropriate procedures for modelling global environmental phenomena. The data used in producing this global analysis were extracted from the Digital Chart of the World (DCW) database, which is drawn from the Defense Mapping Agency's Operational Navigation Charts at 1:1,000,000. Although it remains the principle source of data on access and settlement at global and regional scales, DCW is geographically inconsistent in the level of detail it provides and is rather outdated as some of the ONC charts on which it is based date to the mid-1970's. Therefore, improvement of the spatial data on settlement and infrastructure would be a necessary part of a global assessment of forest naturalness using this approach. Some progress can be made by drawing on the available higher quality national and regional data sets.

Data Scale Issues

The accuracy and spatial precision of any value obtained from a spatial model can be no better than the accuracy and precision of the primary attribute data from which it is derived. This means that the scale of the primary attribute data that are available to an analysis of ecological integrity or naturalness, in effect, represents a limit to the confidence that may be placed on results and their interpretation.

The impact that the accuracy and spatial precision of primary attribute data may have on derived index measures is well illustrated by comparing the results drawn from two analyses of the Australian island state of Tasmania using AWI methodology (Fig. 19). In one case the data are drawn from the global analysis shown in Figure 18, for which the primary attribute data come from DCW. In the other, the data are derived from the Australian assessment, shown in Figure 17, which was derived from a range of very high quality primary attribute data. Most access, settlement and infrastructure data were extracted from local 1:100,000 and 1:25,000 scale topographic mapping. High quality land use and land cover data were utilised, drawn directly from relevant land management and mapping agencies. The Australian assessment was conducted at a grid resolution of 500m.

A comparison of the results of these two analyses shows that both successfully discriminate places with the highest relative wilderness quality. The south-west of Tasmania is notable in this regard as a stand-out feature of both studies. This region is regarded as one of the three key cool temperate wilderness areas of the Southern Hemisphere. Both studies also pick out most other places that are regarded as significant as wilderness within the Australian context.

However, scale difference between these two analyses is evident in the ability of each to discriminate smaller areas of significance as wilderness, and in the accuracy and precision of these assessments. Only the Australian study is capable of identifying areas that may have significance as wilderness in a local or Tasmanian context. Moreover, only the Australian study has sufficient accuracy and precision to be useful for operational evaluation and planning purposes. The global analysis is far too imprecise and incomplete in this respect.

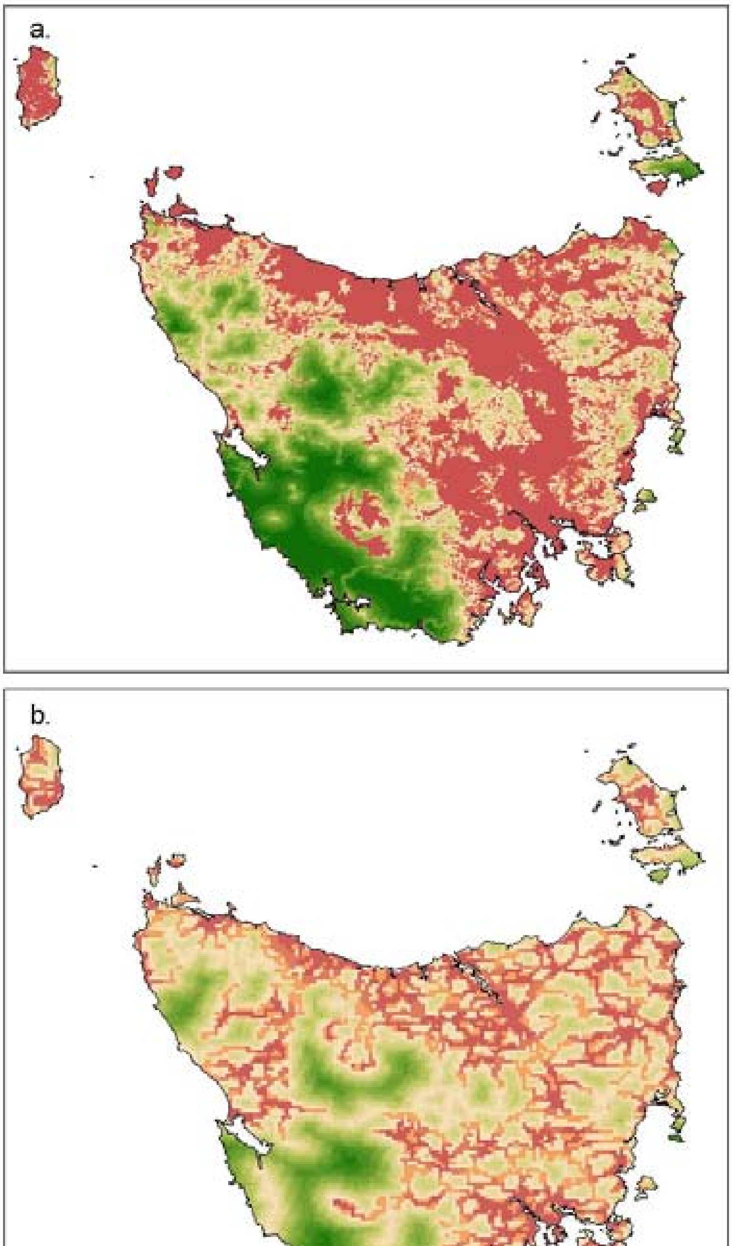

Figure 19. Australian Wilderness Index values for Tasmania, as extracted from (a) the detailed Australian analysis shown in figure 17 and (b) the global analysis (Fig. 18), which was done at a coarse (2.5 km) spatial resolution and included no biophysical naturalness component. Both analyses successfully discriminate places with the highest relative wilderness quality, such as the south-west of Tasmania, which is regarded as one of the three key cool temperate wilderness areas of the Southern Hemisphere. However, the Australian study (a) is strongly influenced by the inclusion of the biophysical naturalness component, which reduces the wilderness values of agricultural and other modified areas, and clearly includes more detailed and comprehensive data on infrastructure and its influence. The Australian study also identifies more explicitly small areas that have higher wilderness values than their surroundings, which may be significant in local or Tasmanian contexts. Only this study provides sufficient accuracy and precision to be useful for operational evaluation and planning purposes.

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Presentation of Results

Bearing in mind the constraints on analysis and interpretation discussed above, this approach remains a promising one for providing insight into forest naturalness at global and regional scales. Once a wilderness surface has been generated, it can be intersected or overlaid with mapped forest cover distribution to assign all forest cover to a wilderness quality or naturalness category (Fig. 20). This can is turn be displayed as Forest Naturalness maps that are analogous to the Forest Spatial Integrity maps shown in Figures 9, 11 and 13. Alternatively, statistical summaries of forest area by naturalness class (cf. Figs 10, 12 and 14) can be generated to provide a baseline for monitoring. The maintenance of forest naturalness within some target range would provide a useful basis for evaluating policy and management relating to forests in the context of forest biodiversity preservation.

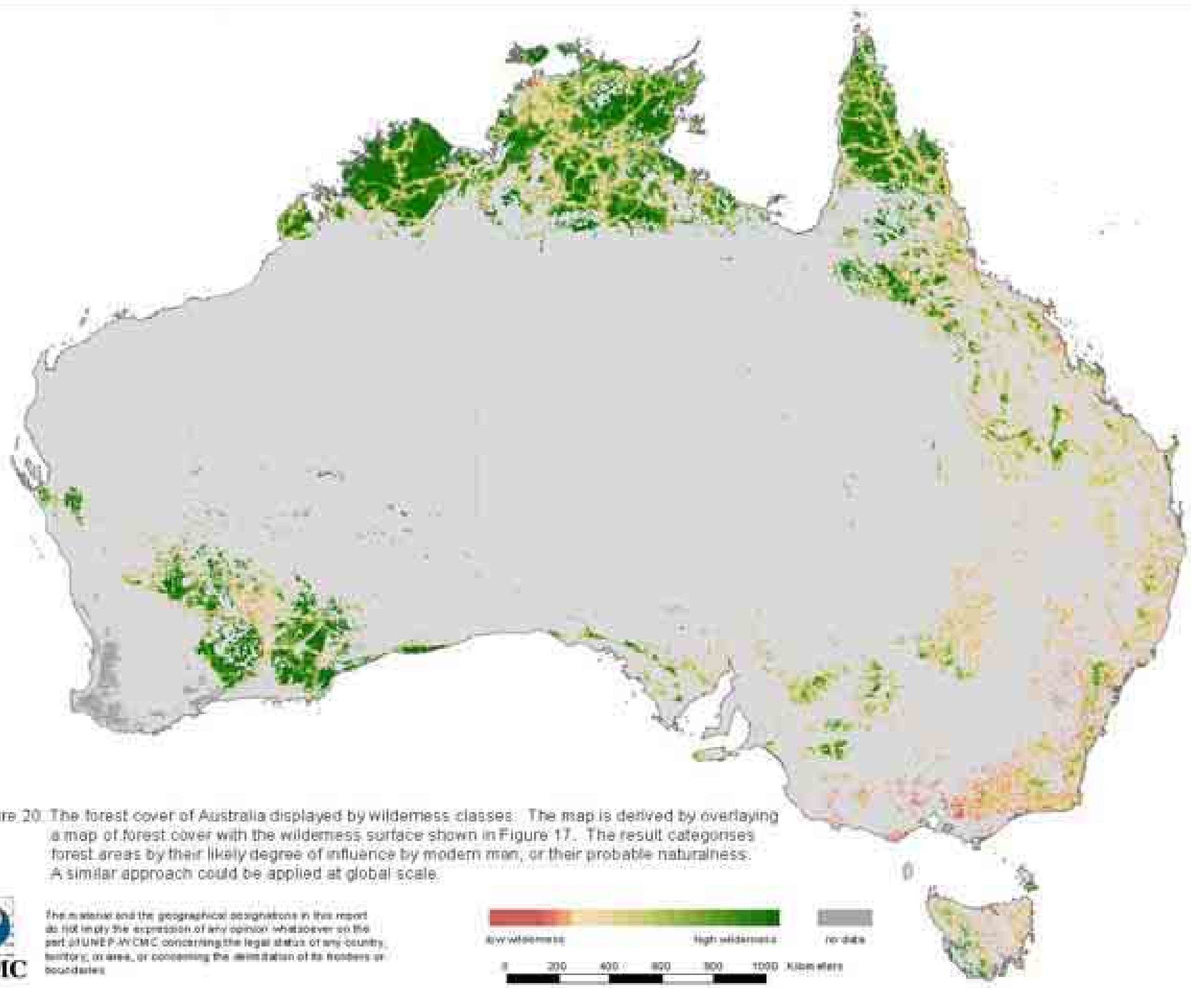

Figure 20. The forest cover of Australia displayed by wilderness classes. The map is derived by overlaying a map of forest cover with the wilderness surface shown in Figure 17. The result categorises forest areas by their likely degree of influence by modern man, or their probable naturalness. A similar approach could be applied at global scale.

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V Conclusion

This paper has presented two approaches that could be applied at global and regional scales to deliver significant advances in assessment and monitoring of the status of forests and their biodiversity. The first of these is the evaluation of forest spatial integrity, which encompasses the size, configuration and isolation from each other of forest areas. Declining spatial integrity of forests is a consequence of many types of human activity, especially land conversion, and is likely to have adverse effects on the natural biodiversity complements of remaining forests.

Other impacts of human activity are less easily measured directly, and can be evaluated better by estimating the likelihood of human influence on the ecosystem. A naturalness indicator, such as the Australian wilderness index, permits the construction of a surface measuring remoteness from human influence that can be overlaid with forest cover data to derive indices of forest naturalness.

Both of these indices can be displayed in mapped form, with each unit of forest cover given its own index value. They can also be used to generate statistical summaries of forest spatial integrity and naturalness that can be used as baselines for monitoring if the same transparent methodologies are applied consistently through time.

The implementation of baseline assessments and subsequent monitoring of forest spatial integrity and naturalness as proposed in this paper would be a significant advance over current practice, and would incorporate biodiversity preservation as one of the multiple benefits of forests included in the periodic assessment of the world's forest resources. This would make it possible to follow trends, not only in forest quantity, but also in its quality with respect to preserving biodiversity.

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