

Forest biodiversity assessment for reporting conservation performance

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ABSTRACT

There is a need to upgrade the quality of information about the status of indigenous biodiversity so that agencies (e.g. Department of Conservation) can make appropriate conservation management decisions. Methods and indicators for determining changes in indigenous forest biodiversity are developed. Because indicators, and the way they are derived, will change over time, it is most essential that any biodiversity assessment system is based on an enduring set of compositional, structural and functional characteristics. Experience in indigenous forests suggests the following are required for monitoring systems: build on past information while accommodating new developments; pay more attention to sampling design; select indicators that achieve goals; do not focus too much attention on today's specific views and concerns; allow for interpretation of indicators in monitoring designs; and, do not expect an indicator will necessarily return to some pre-disturbance value or trajectory. Six indicators are proposed: forest area and fragmentation as a habitat indicator; tree mortality and recruitment rates for maintenance of structural dominants; community composition as an indicator of species assemblages; exotic weeds as a measure of intactness; indices for introduced animal impact; and, quantity and characteristics of dead wood as a habitat diversity indicator. Many of these indicators are currently best assessed through a network of permanent plots. There is also considerable merit in having indicators which can be used in predictive models to develop time-frames for management intervention. These indicators are assessed in relation to other national and international initiatives, including the Biodiversity Strategy. Although this report was specifically commissioned for forests, such a system should eventually be established to cover the full range of ecosystems.

Keywords: Forest, biodiversity monitoring, conservation performance, achievement reporting, forest health, indicators.

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1. Introduction

There is widespread concern that New Zealand's indigenous biodiversity continues to deteriorate because of anthropogenic influences such as the introduction of weeds and pests. The need to protect and report on the status of indigenous biodiversity, both nationally and internationally, is encompassed in legislation (Conservation Act 1986) and international obligations (e.g. Convention on Biological Diversity). Landcare Research, Lincoln, investigated methods and indicators of indigenous forest biodiversity for the Department of Conservation (DOC).

It was agreed that the *investigation* would result in a discussion paper (this report) on methods and indicators for assessing the need for conservation management intervention in indigenous forests at national and regional scales. (Indicators are parameters based on changes in the composition, structure and functioning of natural systems). These *methods* and *indicators* should take account of other research funded by DOC, other national initiatives such as the Ministry for the Environment green package carbon project, and international reporting requirements (see Bellingham et al. 2000). The paper will *recommend* to DOC a system or systems it may establish to address its needs for planning, decision making, and reporting on the status of indigenous forests, and contribute to quality conservation management. The *approach* will optimally integrate existing information to provide maximum continuity of biodiversity information. The *result* will be a clearer understanding of the status of indigenous forests for assessing the national priority conservation outcomes (Department of Conservation 2002) such as:

- No avoidable human-induced extinctions; indigenous populations have long-term security within their natural range.
- Appropriate monitoring systems are designed and implemented.
- A comprehensive and representative range of natural habitats and ecosystems important for indigenous biodiversity is protected.
- Unwanted organisms which pose significant risks to indigenous biodiversity that are newly established or not yet widespread, are eradicated or contained.

These are rather challenging outcomes for DOC. A comprehensive method of assessing biodiversity changes, with a wider focus than simply gains from targeted expenditure, is the only way of truly judging whether such outcomes are achieved. Although this report specifically addresses forests, the monitoring principles discussed can be applied to a wide range of ecosystems.

2. Background and objectives

The Department of Conservation recognises the need to upgrade the quality of its information about biodiversity for appropriate conservation management (Department of Conservation 2002). This need is also recognised in New Zealand's recently published biodiversity strategy (The New Zealand Biodiversity Strategy, Department of Conservation & Ministry for the Environment 2000) which calls for consistent monitoring methods to provide information on key changes in extent and condition of biodiversity. This need is also expressed in paragraphs (a) and (b) of Article 7 of the Convention on Biological Diversity (CBD), which calls for identification and monitoring of biodiversity important to conservation and sustainable use.

Reasons why DOC must collect information about indigenous forests include:

- **To evaluate the effectiveness of management.** The need to manage forests for the protection of biodiversity is enshrined in legislation (e.g. Forests Act 1949, Conservation Act 1987, Resource Management Act 1991). Public and private forest managers require information to evaluate the effectiveness of management decisions and expenditure intended to protect or enhance indigenous biodiversity.
- **To report on biodiversity.** International, national, and regional reporting requirements broaden the scope of information required by DOC and other agencies (e.g. Bellingham et al. 2000). New Zealand is a signatory to several international agreements with specific reporting commitments for forests (e.g. Forest Resource Assessment 2000).
- **To increase the knowledge base.** Changes occur in the composition, structure, and functioning of forest ecosystems at a range of time scales. Quantifying and understanding changes driven by human-related impacts, within a context of natural processes, requires improved data on patterns and rates of change. Increased knowledge is also an important by-product of information-gathering for other purposes, such as those just described above.

This report discusses how past monitoring activities provide principles for designing new monitoring systems, as well as describing methods and indicators that would satisfy a range of needs into the future. The challenge is to design, where possible, a system flexible enough to satisfy multiple needs from monitoring, at a range of spatial and temporal scales. An earlier draft of this report was presented to, and received comments from, a number of DOC staff members (e.g. Frimmel & Turner 1999).

Methods and indicators are proposed that should be useful, from DOC's perspective, for assessing changes in the composition, structure, and functioning of indigenous forest biodiversity at regional and national scales.

3. Lessons from past experience

There has been a long history of methods and systems being developed for measuring biodiversity in New Zealand's indigenous forests (e.g. McKelvey et al. 1958; Wardle et al. 1971; Stewart et al. 1989; Meurk & Buxton 1991; Hall et al. 1991; Wiser et al. 2001). The following examples indicate some of the diversity in approaches that have been taken. Specific manipulations of introduced animal populations, and related vegetation assessment, have often been used to assess the benefits from conservation expenditure at a local scale (e.g. Stewart Island coastal forests, Stewart & Burrows 1989). The status of indigenous biodiversity has been assessed in various ways at local and regional scales, e.g. aerial imagery and permanent plots to assess the extent of forest dieback, and its relationship to possum impacts (Pekelharing & Reynolds 1983; Rose et al. 1992; Rogers & Leathwick 1997; Bellingham et al. 1999a). Forest plots have been used at a national scale to report on tree species composition and structure of forests (Nichols 1976; McKelvey 1984), as well as biomass carbon storage (Hall et al. 2001). It can be seen from these examples that methods and systems useful at a range of scales has been a requirement from conservation managers for a long time.

3.1 WHAT DOES EXPERIENCE TELL US ABOUT SELECTING METHODS AND INDICATORS?

3.1.1 **Build on the past while accommodating new developments**

The recording, measurement, and interpretation of human-related changes in New Zealand's indigenous forests has been characterised by a plethora of techniques. The merits and limitations of each are, to some degree, based on the individual preferences of different researchers, what spatial and temporal scales are relevant, and what interpretations are possible. In some extreme cases, persuasive individuals have caused resources (including large investments of money) to be focussed on specific techniques, which may or may not have been effective. Often such techniques were attractive because of their claims of efficiency and simplicity. Use of a number of these techniques stopped when the individual (or group) ceased to be directly involved with forest monitoring or lost their former influence. What can seem, at times, as 'apparent anarchy' in techniques does not help the long-term need for understanding the changes in our indigenous forests. This is particularly so when the debate about techniques becomes a distraction from achieving a broad consensus on the type of characteristics to measure in forests.

Any change from one technique to another is likely to limit the usefulness of data because of the long time frames associated with data collection for monitoring (see comments in Hutcheson et al. 1999). Our aim must be to assure comparability of data over decades, perhaps centuries: this provides a challenge in many respects. One overriding principle to achieve compatibility and, hence,

long-term records, is to always build on the past. This does not necessarily mean we must continue to use the same techniques, but at least the succession of techniques must be explicitly comparable so that we are not always starting at time zero. Another issue is quality assurance over time, and comparability of data. Achieving this requires standardised techniques that are carefully described in field manuals linked to a data management system that stores the raw data in a specified and well-documented way. A system providing such a structured approach to measuring changes in indigenous forests nationally is included in the field methods described by Allen (1992, 1993) that are linked to the NVS (National Vegetation Survey) databank (e.g. Hall et al. 1991; Wiser et al. 2001). These methods have underpinned the measurement of biodiversity patterns and dynamics in many localities throughout New Zealand over recent decades (e.g. Stewart & Burrows 1989; Smale et al. 1995; Burrows et al. 1999).

3.1.2 Sampling design

For evaluating conservation performance and national-level reporting it is important to select a system that records indicators in an unbiased way. Although some methods can provide complete coverage of the whole country (e.g. satellite imagery), others require an approach that samples forests in a representative way. For plot systems, the options are random or systematic location of plots throughout the forests nationally or within pre-determined strata. It is worth including here some details on the design of the Forest Health Monitoring Program used in the USA, which samples that country's forests in an unbiased way (summarised from Stapanian et al. 1998; USDA Forest Service 1998). The design permits the estimation of indicators at state, regional and national levels with known levels of confidence. It has been determined that combining site-specific studies in particular environments is not appropriate for rigorous testing at these levels (Stapanian et al. 1998). The design chosen for the monitoring system uses an equal-area triangular grid with points approximately 27 km apart. The country was tessellated in this manner with hexagons. Within each hexagon, a plot is located at the hexagon centre. The design is sufficiently flexible to accommodate post-stratification and aggregation into, for example, eco-regions (Rogers et al. 1998).

Taxa that occupy only a limited geographical range, few types of habitats, or are rare as individuals, will not be covered cost-effectively by a national- or regional-scale design. Threatened species, for example, along with local conservation issues (e.g. sika deer in the Kaweka Range) or intensive management (e.g. mainland islands), require intensive local-scale monitoring beyond that provided for by a national or regional system (e.g. Stanley et al. 1998). The contribution a national or regional system may make to such issues is in exposing new locations for some rare species, and signalling, over time, when some declining species become rare and require more intensive assessments. There will always be a need for local-scale assessments. Bellingham et al. (2000) present a three-tier option for a network of permanent plots in New Zealand:

- Tier one—spatially extensive, non-stratified, representative national network using some existing plots.
- Tier two—local networks using some existing plots, but expanded to address geographic deficiencies in tier one sampling schemes (suitable for Conservancies).

- Tier three—detailed, multidisciplinary, long-term ecological research sites (e.g. some mainland islands).

Although this report focuses on the national scale, these three tiers need to be designed in concert and integrated across the three scales for all types of land cover.

New Zealand is well endowed with permanent monitoring plots in indigenous forests, and one indicator for which there has been a rigorous testing for bias is a NVS plot-based national estimate of carbon storage (Allen et al. 1998). That project, funded by the Ministry for the Environment, developed a set of criteria for selecting an existing plot nearest a systematic set of sampling points located throughout the country's indigenous forests. The sampling points for a 1990 estimate were on a 9 km × 9 km grid and resulted in a live biomass carbon estimate that was unbiased in terms of elevation, climate, forest, and soil types as well as plot density (Hall et al. 2001). There was some geographic bias, however, because there were few permanent plots in some extensive areas of forest (e.g. inland Taranaki).

It is important to sample the geographic distribution of forests because of, for example, regional differences in disturbance regimes, which may or may not relate to environment. It was for such reasons that the Ministry for the Environment project decided for subsequent carbon estimates to use a systematic grid system of plots rather than any form of stratification (Fig. 1; Allen et al. 1998; Coomes et al. 2002). Grid systems are often desirable over random locations because they give better spatial coverage. As Bellingham et al. (2000) have stressed, it is important to incorporate existing plots with long-term data in the evolution of any design; some researchers give this a higher priority than particular statistical considerations. So, some of the grid points in Fig. 1 have an existing forest plot that can be used to represent the point, while others require that the carbon monitoring system establish a new plot. Any interpretation related to specific factors influencing composition, structure, and function of forest ecosystems would then be made subsequently.

Some authors consider that any form of forest stratification is inappropriate where the goals are long-term assessments of forest at national and regional scales (e.g. Overton & Stehman 1996; Bellingham et al. 2000). It is unlikely that any particular form of pre-stratification (e.g. administrative boundaries, forest types, environment types, catchments, disturbance levels) will adequately meet a wide range of purposes and therefore endure in the long term (e.g. decades and beyond). It is better for a monitoring system to be implemented with few such constraints. Overton & Stehman (1996) considered that the statistical advantages from a simple design will, over time, compensate for possible precision or efficiency lost by not using more complex designs. It is for such reasons that many countries (e.g. USA, Finland, Switzerland, Canada and Austria) employ grid-based systematic designs.

So far this section has considered the *location* of sampling points to give an unbiased estimate of an indicator nationally. The second aspect to be considered is *how many points* are required for a given level of precision. There have been few attempts to generate the sampling intensity required to achieve a given level of precision for indicators in New Zealand indigenous forests, either locally or nationally (but see Bachelor & Craib 1985; Bellingham

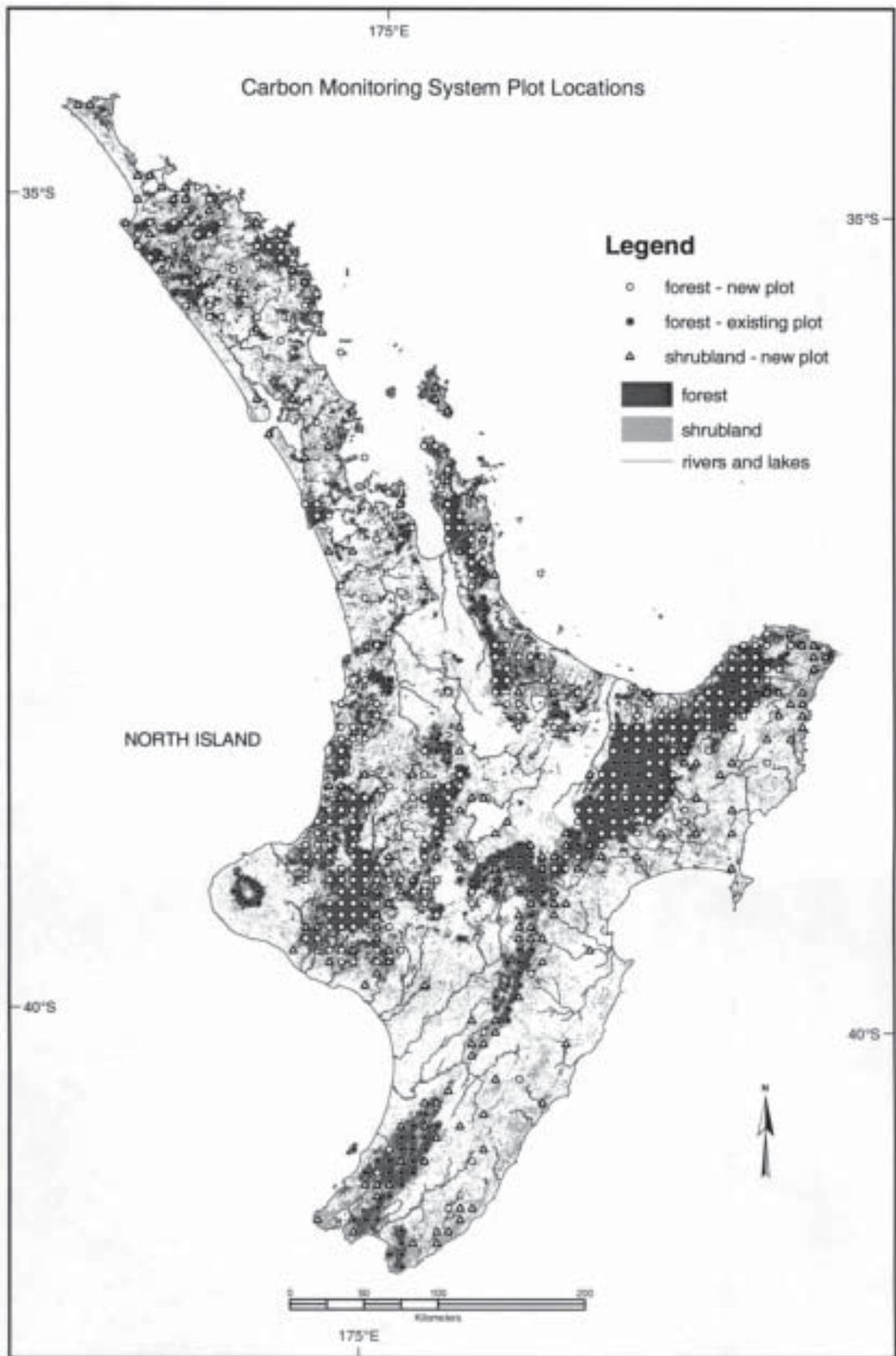
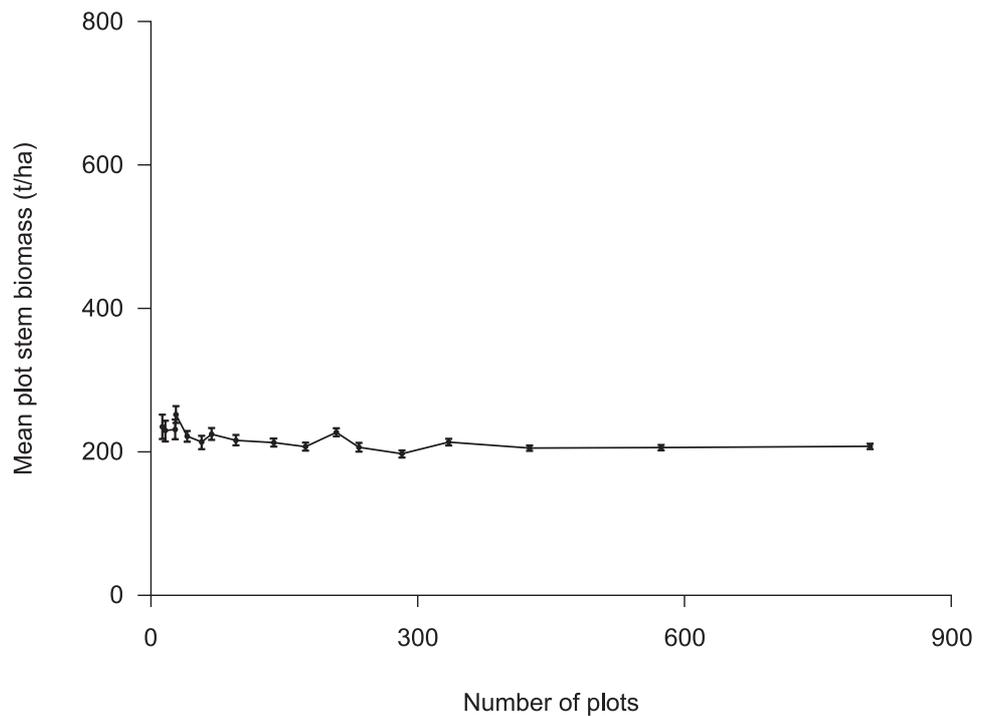


Figure 1. An 8 km × 8 km grid giving carbon monitoring system plot-sampling points superimposed on North Island forest and shrubland areas from the Land Cover Data Base of New Zealand. Whether an existing plot appropriate for monitoring carbon is associated with a grid point is also indicated (map adapted from Allen et al. 1998).

Figure 2. Mean plot stem biomass, and associated standard error, for various plot densities nationally (after Hall et al. 2001).



et al. 2000). For example, Hall et al. (2001) showed that approximately 570 plots throughout the country are required to calculate a plot-level mean biomass carbon estimate to within 5% (at a 95% probability level). The relationship between the mean plot biomass (and associated standard error) versus number of plots is given in Fig. 2. Coomes et al. (2002) have shown that re-measurement of plots would allow analyses with repeated measures, and reduce the number of plots required to detect a 5% change in carbon storage to about 200 nationally. This sampling intensity indicates what is required nationally; any reporting from smaller spatial scales would require a similar set of analyses. Clearly, any reporting of conservation performance will require that sampling intensities be determined for the range of selected indicators at a specified level of precision, for the range of spatial scales for which they are used. We must also establish how often sampling points should be re-measured. The existing NVS database could provide much of the data necessary to test such sampling design issues at a range of spatial scales.

3.1.3 Selected indicator must achieve goals

In the past, it has often been the case that conservation goals were not sufficiently defined to enable performance evaluation in relation to those goals. DOC's guidelines for measuring conservation projects address this issue, and should lead to an improved outcome for quality conservation management and acceptance by conservation managers (Department of Conservation 1998). Within this report, the characteristics of interest are changes in forest composition, structure, and function, and how these are affected by human-related impacts. At any point in time these characteristics can be used to derive indicators of biodiversity status. Because indicators, and the way they are derived, will change over time, it is most essential that any biodiversity assessment system measures an enduring set of characteristics. An indicator is derived from measurements showing changes in forest composition, structure,

and function. This would focus the selection of such indicators away from human-related pressures such as atmospheric pollutants in wet deposition, the number of tourists walking the Lake Waikaremoana track, the extent of stoats in forests, or the number of introduced herbivores, such as deer. It must be remembered, however (as discussed in 3.1.5), that it will sometimes be necessary to quantify the level of these pressures, to correctly interpret human-related impacts.

There is an inherent tendency to view an increase in a particular indicator as a desirable outcome for conservation performance. However, this is a naive approach. For example, in a study of ecosystem properties in forests inside and outside deer exclosures, values of humic carbon and nitrogen were sometimes greater in the presence of deer (Wardle et al. 2001). Similarly, surveillance monitoring that shows no change in an indicator is, in itself, a useful result. Reporting data that only show change biases the representation of processes that are occurring. This assumes, of course, that there is an adequate statistical design.

3.1.4 What is a suitable indicator?

It is very desirable to have specific indicators that can be clearly interpreted. It is also desirable that each indicator can be used for, and partitioned in terms of, more than one human-related impact. In reality, our knowledge about what controls the composition, structure, and functioning of forest ecosystems is at a stage where the indicators we can select today fall well short of these expectations. Taking this into account, are there some principles that can be followed when selecting indicators? The following general criteria have been suggested for the selection of biodiversity indicators (Pearson 1996):

- Species selected must be taxonomically well known and stable
- Their biology and life history must be well understood
- They should be taxa that occupy a breadth of habitats and geographical range
- They should also be taxa where there is specialisation of populations within a narrow habitat
- The temporal patterns reflected in the indicator taxon should be reflected in other taxa
- They should have potential economic importance

These criteria do not emphasise taxa that are of specific concern now; current concerns should be encompassed within the criteria (see Hutcheson et al. 1999). This approach should apply to the full range of biodiversity indicators. New Zealand's indigenous forests have been quantitatively classified based on composition at various spatial scales, including nationally. One national indicator may be the area of each forest type; however, at another point in time it may be more useful for some purposes to classify the forests by level of disturbance. Taking a specific example: DOC is, at a regional scale, most interested in classifying Kaweka and Kaimanawa forests by natural canopy disturbance so that monitoring activities can be focused on whether tree species are regenerating in open-canopy areas with or without deer control.

The advantages of establishing a comprehensive monitoring system that does not focus solely on a current issue can be shown by the following example. In the 1970s there was concern about the impacts of red deer (*Cervus elaphus scoticus*) on regeneration of eastern South Island beech forests. A quantitative

study involving permanent plots was set up in the Harper-Avooca catchment to investigate deer impacts (see Section 3.1.5). At the same time, ecologists and land managers were becoming concerned about the invasion of eastern South Island tussock grasslands by exotic *Hieracium* species (Rose et al. 1995). Many consider this invasion to be a consequence of land degradation brought about by repeated burning and sheep grazing. Fortunately, the permanent plots set up in the Harper-Avooca forest to monitor deer impacts on tree regeneration also recorded exotic plant species. This dataset shows a marked invasion of beech forests by *Hieracium*, which calls into question the land degradation paradigm (Wiser et al. 1998). The message here is: if characteristics are selected that are too specific, and focussed on today's concerns, we will be constrained in the future. We also do not want 'time zero' for the indicators derived to always be the date at which an issue they reflect became a major concern. An indicator system that provides most options for the future is one that has clear links to issues of concern but which is also explicit about, and minimises, today's prejudgements.

3.1.5 What does change in an indicator mean?

The structure, composition, and functioning of forest ecosystems correlate with many factors (Table 1); some of which are relatively easy to determine (e.g. precipitation) and others that we know little about (e.g. impact of infrequent disturbance impacts). Additionally, because many of these factors can be inter-correlated, at any particular scale, it is difficult to separate the impact of any individual factor and define causal relationships. This results in limitations to the interpretation of spatial and temporal variation in indicators.

The following example highlights spatial limitations. Mortality of some tree species has commonly been used as an indicator of brushtail possum (*Trichosurus vulpecula*) impacts. A previous study in south Westland evaluated tree mortality in three localities stratified along a possum invasion front; it was only in the write-up stage of the study that the invasion front was shown to also coincide with a soil fertility gradient related to plant composition (Stewart 1992). Because we know that the frequency of dead trees in such forests apparently

TABLE 1. FACTORS COMMONLY INFLUENCING THE DISTRIBUTION, STRUCTURE, COMPOSITION, AND FUNCTIONING OF FORESTS. SOME OF THE VARIABLES COMMONLY MEASURED TO REPRESENT THESE FACTORS, AS WELL AS SOME OF THE MECHANISMS THROUGH WHICH THESE FACTORS MAY OPERATE, ARE GIVEN.

FACTORS	VARIABLES	MECHANISMS
Disturbance	changes in biomass or number of trees	individuals killed of one or more species
Herbivory	level of defoliation, individual height growth	reduced photosynthetic ability
Species traits	litter quality and mix	modifies the ability of seeds of other species to germinate
Climate	temperature, precipitation	changes physiological processes
Soil	texture, N availability	influences resources essential for growth and development
Dispersal	seed dispersal, available regeneration niches	seeds do not arrive at an otherwise suitable site
Time	tree age, relative biomass	species differential longevity

relates to a soil fertility gradient (see Allen et al. 2002), it was difficult to partition the effects of a possum density gradient along the Westland invasion front from those driven by soil fertility. This study demonstrates how the interpretation of indicators can be compromised by pre-stratification: in this instance, because of unknown correlations of variables at the time the study was set up. Overcoming such limitations will be enhanced over time by the increasing amount, and spatial resolution, of complementary information (the factors shown in Table 1).

It is easy to put forward a set of indicators, but experience tells us that it is usually rather more difficult to interpret any changes in a particular indicator unambiguously and, therefore, what is indicated. The ideal solution would be to select indicators for which there is only one explanation for any changes. Such indicators appear to be the basis of simple relationships between introduced animal densities and their impact. For example, a deer impact model has been proposed, which relates deer density to seedling growth for plant species varying in palatability (e.g. Department of Conservation 1997). Although such models accept that variability in the relationship may be explained by other factors, they do not account for the fact that other parameters co-vary with animal density. Figure 3 presents a series of graphs useful for clarifying this point. The data comes from the Harper-Avoca catchment where deer density (frequency of defecations), tree regeneration (mountain beech seedling density), and overstorey stocking (basal area) were monitored in mountain beech (*Nothofagus solandri* var. *cliffortoides*) forest (Hickling 1986; Harcombe et al. 1998; Allen unpubl. data).

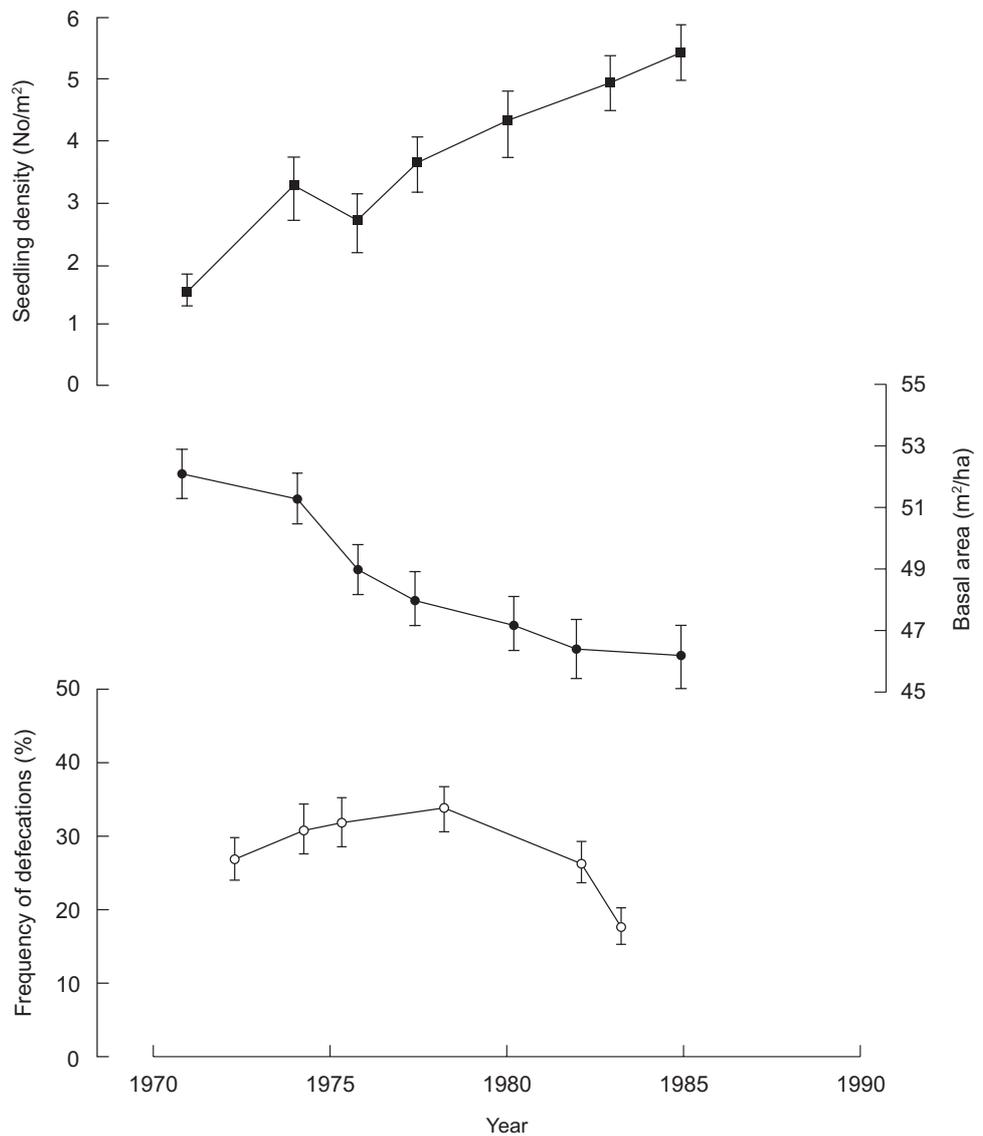
If we first consider the relationship between deer density and regeneration between 1978 and 1984, we might conclude that seedling density increased in response to a decline in deer density. On the other hand, between 1973 and 1978, we might conclude the converse, that seedling density increased as deer density increased! Only with the additional measurement of tree basal area is it clear that seedling density more than likely bears no relationship to deer density but is, instead, responding to declining tree basal area. Elsewhere, deer apparently do effect regeneration of mountain beech (e.g. Allen & Allan 1997). The point being made here is that for strong interpretation of changes in indicators, it is important to also collect carefully selected complementary data over the same period, rather than to assume that simple cause-effect relationships exist.

3.1.6 What is the resilience of forest ecosystems?

A common expectation is that reducing human-related impacts will result in an indicator based on forest composition, structure or function returning to some pre-impact or baseline level. This expectation has often influenced the selection of indicators. It is a challenging exercise to reconstruct a baseline, unmeasured in any detail before human-related impacts. A more realistic view is that forests will continue along a dynamic trajectory reflecting history—and conservation management needs to decide if that is a desirable trajectory.

An example can be used to explain this issue. Extensive synchronous mortality of southern rata (*Metrosideros umbellata*) and Hall's totara (*Podocarpus hallii*) has arguably been attributed to browsing by brushtail possums in Westland conifer broadleaved-hardwood forests (Rose et al. 1992; cf. Veblen & Stewart

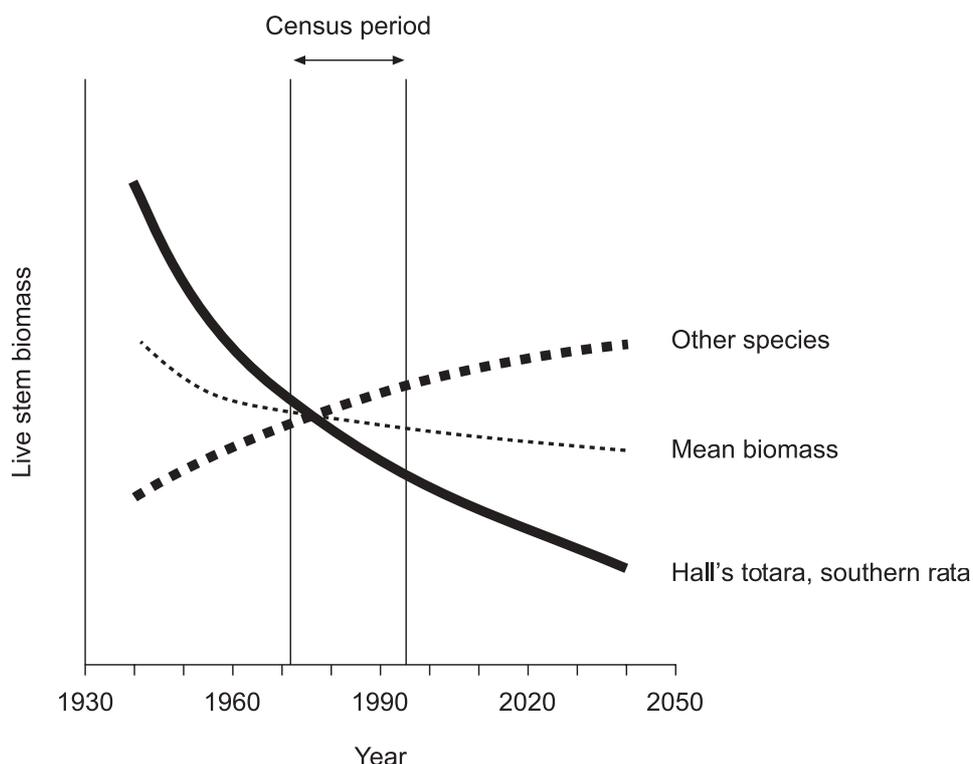
Figure 3. Relationships between deer density, tree basal area, and seedling density between 1970 and 1985 in the Harper-Avooca catchment, Canterbury.



1982). Although there has been a decline in the stem biomass of these two species over the period of quantitative measurement, it is unclear what the initial levels of dominance in these forests were (Fig. 4). It is also likely that there has been considerable variability over recent centuries in the importance of southern rata and Hall's totara in these forests. This variability would have been related to disturbance events such as earthquakes (Wells et al. 1998). Therefore, a baseline selected from the past will not reflect the background dynamic state of these forests and where they would be in terms of species dominance today, without the impacts of introduced herbivores.

To further develop this Westland example, in some places where there has been complete loss of southern rata and Hall's totara from the forest canopy, there is a paucity of regeneration by these species (Allen & Rose 1983). Regeneration may be viewed as an early indicator of forest recovery. Allen & Rose (1983) suggested regeneration limitation was a consequence of the lack of seed input to large dieback areas. There are alternative explanations. In Hawaii it has been shown that certain litter types can inhibit *Metrosideros* seed germination (Walker & Vitousek 1991). Increasingly, studies are showing that effects related to individual plant species are important determinants of ecosystem properties,

Figure 4. Mean live stem biomass of all species (mean), Hall's totara, southern rata, and others, for the Kokatahi catchment (from Bellingham et al. 1999a). The relationships are based on permanent-plot measurements (over census period) and subjective reconstruction.



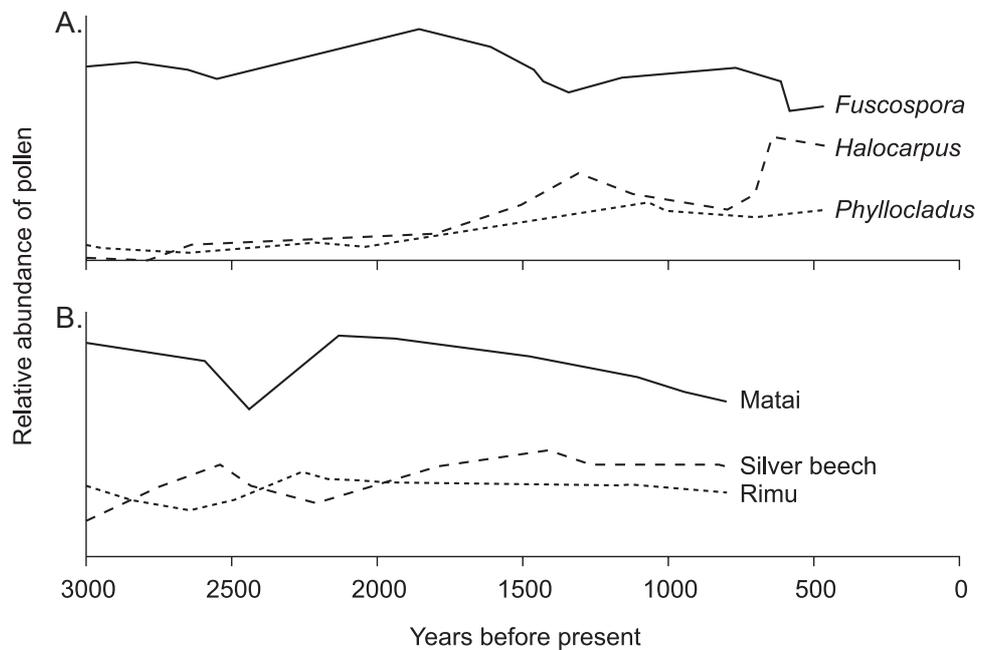
and that these effects may over-ride the importance of abiotic factors (e.g. Wardle et al. 1998). In addition, these effects on ecosystem properties (e.g. soil biota and nutrient mineralisation) will likely have important consequences on the nature of trajectories following human-related disturbance. Because of the long-lived nature of many tree species in Westland forests (e.g. 800 years for southern rata), a return to the pre-disturbance forest structure is at least a millennium away—assuming such a trajectory is actually possible.

There is no reason to expect more stable, or known, baselines at longer time scales. Figure 5 shows the relative abundance of tree species, based on pollen abundance, at two locations over several millennia, prior to human-induced deforestation. What should we select as the baseline? Immediately prior to deforestation or 2000 years before present? Both would give very different baselines. Because of the natural variations in forests over time, there is no reason to expect that an indicator of ecosystem properties will necessarily return to some pre-disturbed state. The essential point here is that forest managers will need to decide on what are desirable services and features of forests—which may not reflect historical conditions—and manage to achieve these against a background of change.

In summary, lessons from the long history of monitoring New Zealand forests include:

- Data must remain explicitly comparable over time as techniques evolve
- Data collection must not focus only on contemporary perceptions and needs
- Complementary interpretive data must be collected
- Establishing baselines is difficult

Figure 5. Changes in forest structural dominants over recent millennia based on pollen abundance prior to deforestation at two localities in the South Island: (A) Mount Horrible, inland Canterbury (Moar 1971) and (B) Merrivale, coastal Southland (McGlone & Bathgate 1983).



3.2 WHAT TYPES OF INDICATORS ARE BEING USED IN OTHER COUNTRIES?

It would be a large exercise to review comprehensively the forest monitoring work being done in other countries. For example, Finland alone has undertaken nine national forest inventories since the 1920s and currently uses satellite imagery, digital map data, and systematically located ground-based plots to provide information on timber stocks and growth, forest health, understorey composition, and bryophyte chemistry for pollution monitoring (Tomppo 1998). The USA has a programme specifically focussed on forest health monitoring with a goal of developing and implementing a co-operative multi-agency programme to monitor, assess, and report on the status, changes, and trends in forest ecosystem health and sustainability (Stolte 1997). In the USA system, satellite monitoring provides evaluations of forest fragmentation and use, aerial monitoring is used to detect local problems, such as pest outbreaks, and ground-based plots provide data on forest structure, diversity, and other site-specific indicators. Indicators of forest ecosystem components and processes relevant to forest health used or tested in the USA are listed in Table 2.

Data from the USA forest health monitoring system now allow, for example, the large-scale quantitative evaluation of exotic plant invasions in forests and its relationship to human disturbance (Stapanian et al. 1998). Such indicators can be used in comparisons to determine the influence of timber harvesting on forest ecosystem components and processes (Keddy & Drummond 1996). It is useful to review lists of indicators used elsewhere because they may contribute to the breadth and type of indicators that could be used in New Zealand. The USA forest health monitoring system was developed to address national concerns as well as meeting international requirements such as the Montreal Process (see Appendix 3). The specific national concerns reflect, in particular, the ongoing impacts of atmospheric pollution which, at least so far, is not of wide-scale concern in New Zealand's indigenous forests. New Zealand has its

own set of specific concerns, which are reflected in DOC's goals of retaining natural heritage values. Currently, these concerns are dominated by the impacts of introduced herbivores, predators and weeds, but also include some global issues such as the potential impacts of climate change.

TABLE 2. FOREST HEALTH MONITORING INDICATORS CURRENTLY IN USE, OR THAT HAVE BEEN TESTED, WITHIN THE UNITED STATES AT NATIONAL AND REGIONAL LEVELS (STOLTE 1997).

IN USE	TESTED BUT NOT YET IN USE
tree species diversity	understorey species
tree regeneration	lichen communities
tree mortality	foliar chemistry
tree growth	dendrochemistry
tree damage	woody debris
tree crown condition	soil carbon
pollution indicator plants	
soils	

4. Proposed methods and indicators

As part of a project assessing DOC's requirements for a national network of permanent forest plots, Peter Bellingham asked DOC Conservancy, Regional, and Head Office staff for their views on attributes that need to be measured. Attributes to be measured in any forest monitoring system were considered by some to 'depend on the specific questions being asked'. There was no clear pattern among those canvassed as to what are desired attributes to be measured in a forest monitoring programme. Opinions ranged from advocating comprehensive monitoring, to targeted monitoring (e.g. presence or absence of certain forest species; implicit in this is that a causal mechanism is known that explains presence or absence). Variations and inconsistencies in approach were obvious, even within individual responses. For example, one respondent considered that complete measurements of existing permanent forest plots (that sample several stages of tree life histories) may not be necessary, while at the same time advocating the need to additionally measure, at each plot, leaf litter, invertebrates, soils, epiphytes, herbivory effects (by both native and introduced animals), weed invasion, and environmental variables including rainfall, temperature, and UV light. Another respondent considered there would be merit in establishing a nationwide monitoring of forest phenology, while accepting that such a programme would be time-consuming because of the need to conduct monthly observations. Some respondents considered a forest monitoring system should be designed principally to assess effects of vertebrate pests (mainly herbivores), and one advocated a factorial design with various treatments of vertebrate herbivores, while another thought a comprehensive range of attributes measured widely in space and time should enable one to partition the effects of herbivores from natural processes.

From DOC's perspective, useful indicators are those which are relevant primarily to evaluating conservation performance and those which are key to indicating fundamental shifts in the composition, structure, and functioning of forest ecosystems, particularly where these are a consequence of human-related activities. We need to understand why an indicator measured in a certain way is a useful, repeatable approach, and what needs to be measured to help interpret an indicator. The following indicators are considered from these perspectives to meet regional, national, and international requirements.

4.1 FOREST AREA

We must go back approximately 15 000 years to find a time when the New Zealand landscape was as deforested as it is today. A climatic warming saw the dramatic recovery of forests from 15% of the land surface 18 000 years ago to 80% of the land surface 10 000 years ago (Fig. 6). Although natural disturbances (e.g. volcanic eruptions) have periodically removed forests over extensive areas, the forests have usually recovered rapidly. Over the last millennium, human impacts, largely brought about by the extensive and repeated use of fire, have eclipsed the extent and rate of change in forest cover over the previous 20 000 years (McGlone 1989). There has been some recovery of forested area over the past century in areas with a moist, mild climate. The apparent low resilience of New Zealand forest species today is partly a consequence of ongoing disturbance requiring energy input by humans, such as scrub

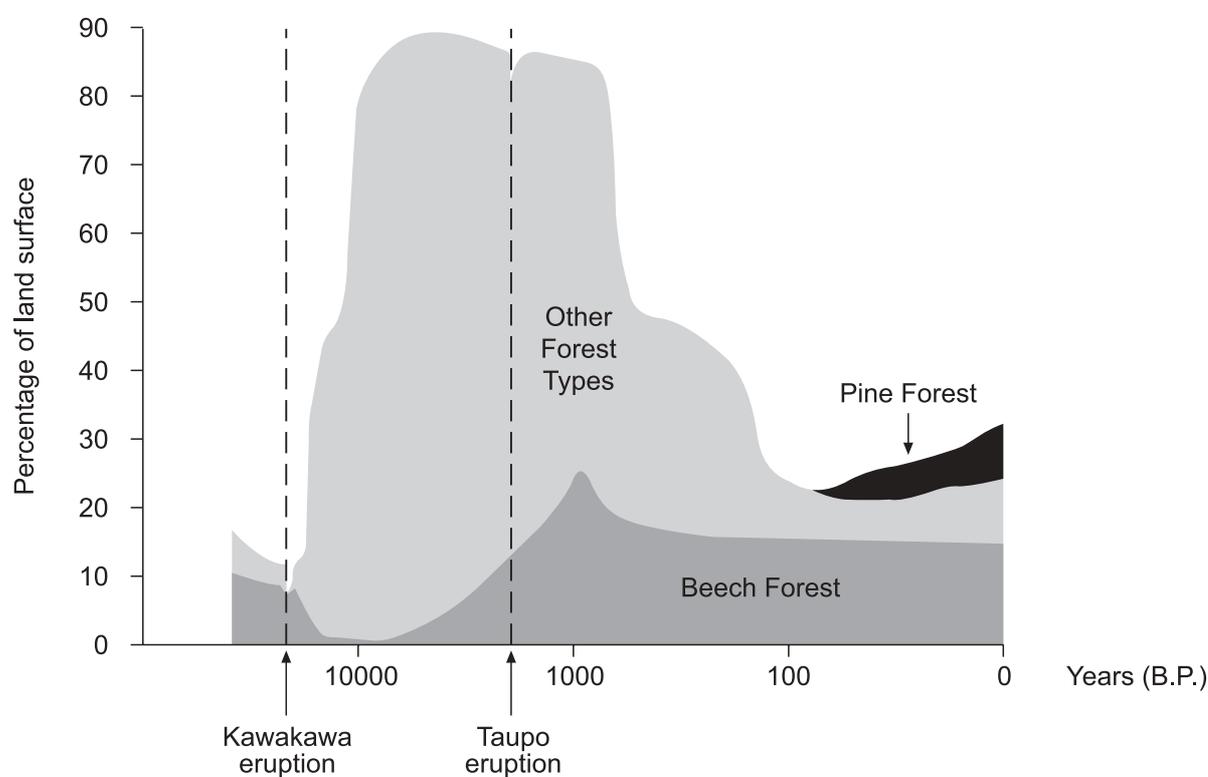


Figure 6. Forested area (percentage of New Zealand) over the last 25 000 years (logarithmic scale). The percentage of New Zealand's area is broken down to beech forest, pine forest, and others. Compiled in consultation with M. McGlone.

clearance. Some losses of indigenous forest area are still occurring. However, with the increasingly marginal economics of extensive, low-intensity agriculture, there is the potential for an expansion of indigenous woody communities; and because New Zealand is naturally a forested landscape, much of this land will eventually develop towards forest. McGlone (1999) argues that, correctly managed, this potential expansion is a major opportunity for the conservation of biological diversity in New Zealand at a landscape scale.

The low resilience of New Zealand's indigenous forests is also a result of modification of fundamental forest processes that has occurred since humans arrived in New Zealand. In the past, indigenous conifer species have rapidly invaded deforested areas, including those dominated by grasses. It appears that establishment by some indigenous conifers (e.g. *Phyllocladus*) is now limited by seed dispersal in some areas. Fragmentation of forests has restricted the migrational opportunities of some species. Disturbance regimes have also been modified, e.g. hydro-electricity development in major river systems has modified flooding regimes and their impacts, for example, on regeneration dynamics of tree species. As a result of such changes, the compositional, structural, and functional characteristics of 'new' indigenous forests may not be the same as those that would have occurred on the same sites without human-related removal of forest.

It will become important to quantify the changes in area, and spatial distribution, as well as the biodiversity within these 'new' indigenous forests as they develop. There has been a history of mapping the spatial distribution of land cover types in New Zealand with complete national coverage being given by the Vegetation Cover Map (VCM) of New Zealand (Newsome 1987). The Land Cover Database (LCDB), based on satellite imagery, gives a greater, and better defined, spatial resolution of forests than the VCM. The LCDB maps indigenous forest (woody vegetation greater than 6 m tall, and greater than 20% cover) distribution at a 1-ha scale. In undertaking the MfE carbon project, Coomes & Beets (1999) have shown that the LCDB misclassifies what is forest about 10% of the time, based on field assessments in the central South Island. The 1-ha resolution of the LCDB allows indices of forest fragmentation, such as size, isolation, shape, and edge:area ratio (e.g. see Norton 2000) to be determined in a way that has not been possible to date, and it can be done at a range of spatial scales. No doubt our abilities to derive landscape-level dynamics will improve with time, but we must ensure compatibility with past techniques, so that trends can be determined. Field data will be required to quantify the compositional, structural, and functional characteristics of 'new' forest communities, as outlined in the following subsections. The distribution of forest with these characteristics may then be generalised by using other types of data at national or regional scales, such as environmental domains or community functional types derived from current and future versions of the LCDB.

4.2 TREE MORTALITY AND RECRUITMENT

Extensive death of canopy trees is a prominent feature in many forest areas (e.g. Ogden et al. 1996). While some canopy disturbance events, such as earthquakes, are unrelated to human impacts, others (such as fire) may be a direct consequence of human activity. However, the separation of human-related canopy disturbance can be problematic because:

- Sometimes the disturbance agent is just not known. For example, there has been a long debate in New Zealand over the level to which possums defoliate and kill trees versus other factors driving the same mortality (e.g. Veblen & Stewart 1982; Bellingham et al. 1999a); or,
- The degree to which some known disturbance agents are a consequence of human activity is unclear. For example, the level to which storminess is a consequence of human-induced atmospheric changes is not known.

The significance of this point is that it identifies a difficulty for forest management focussed on human-related impacts. Extensive animal control operations have often been mounted over the last 40 years where tree mortality is considered to be a consequence of browsing by brushtail possums. This has focussed monitoring for conservation performance on tree mortality. From a wider perspective, because trees are the dominant biotic structural component in forests and drive many of the compositional and functional changes in forests, it is important to measure and understand patterns of tree mortality at a range of spatial scales.

Early accounts describing the existence of extensive tree mortality were based on anecdotal and photographic records (e.g. Grant 1984). Ground-based and aerial surveys have been used to describe and quantify the intensity and spatial distribution of mortality (e.g. Bachelor 1983; Pekelharing & Reynolds 1983; Rose et al. 1992; Rogers & Leathwick 1997). Recognising that forest structural attributes themselves may be important in explaining the intensity and nature of tree mortality, ground-based dead tree measurements have been used to characterise the size-structure of trees that die (e.g. Allen & Rose 1983). However, any point-in-time measurement of dead individuals, irrespective of the method used, will miss an important component of the mortality dynamics: the rate of tree death. Some tree species, e.g. southern rata, likely remain as dead standing spars for more than half a century, while others, such as kamahi (*Weinmannia racemosa*) may decay and disappear within a decade or two (Bellingham et al. 1999a). Ideally, an indicator of tree mortality should be expressed as a rate based on measurements of individuals, and the interpretation of tree death would be aided by a knowledge of the size of individuals and overall stand structure (Hutcherson et al. 1999). In the longer term, an expectation may be that we can model spatial and temporal variability in tree mortality, rather than measure it. However, some of the drivers of tree mortality, e.g. earthquakes (Allen et al. 1999), are so poorly understood in terms of the intensity and spatial distribution of their impact that we will be faced with actual measurement for some time yet.

The maintenance of tree species can also be assessed through regeneration. Tree regeneration is a complex process, as it is influenced by flowering, seeding, dispersal of propagules, seed predation, the availability of suitable

regeneration sites, competitive interactions as juveniles develop, as well as the impacts of introduced browsing animals. Veblen & Stewart (1982) argued that New Zealand ecologists and land managers have a preoccupation with explaining regeneration patterns in terms of browsing animal impacts. Counting seedlings, within size classes, over a fixed area is a useful way of measuring regeneration potential, but any interpretation is dependent on accounting for, at the very least, overstorey conditions and levels of browsing pressure (e.g. Allen & Allan 1997). Seedling counts offer a static view of dynamic situations, and a more useful indicator is one that describes the dynamic nature of the regeneration process.

The maintenance of tree species as components of forest biodiversity is essentially a population process and can be described by, and modelled from, whether tree mortality rates exceed recruitment rates (Bellingham et al. 1999b). Bellingham et al. (1999b) have examined the imbalance of mortality and recruitment of dominant tree species in 14 localities around New Zealand using tagged individuals on permanent plots. Mortality rates of kamahi exceeded recruitment rates by 3.2% per year in Pureora, but elsewhere recruitment rates were generally low and were either similar to mortality rates or slightly exceeded them (Table 3). Mortality and recruitment rates of silver beech (*Nothofagus menziesii*) were generally similar at most localities, with the greatest discrepancy at Mt Arthur where recruitment rates exceeded mortality rates by 0.8% per year (Table 3). While recruitment rates of mountain beech were similar to, or slightly exceeded, its mortality rates in most sites, mortality rates of this species greatly exceeded recruitment rates in two montane forests where it was mono-dominant (by 0.9% per year in Craigieburn and 1.1% per year in Kaweka). Mortality rates of Hall's totara, a minor component of six forests, were higher than recruitment rates in four localities (Table 3). Most notably, 3.0% per year mortality of this conifer at Kokatahi over 23 years contrasts with nil recruitment over the same period, with a less extreme contrast between 0.4% per year mortality and nil recruitment over 21 years in the Caples Valley, and a disparity between 1.4% per year mortality and 0.2% recruitment over 27 years in the Whitcombe Forest. For such monitoring purposes, permanent plots must be of sufficient size to adequately sample the tree populations of various species. Sweetapple & Burns (2002) recently advocated plots as small as 5 m × 5 m for monitoring forests, even though research in most forest types would suggest this size is too small to adequately characterise tree dynamics.

There is accumulating evidence of the highly variable nature, both spatially and temporally, of tree mortality and recruitment. The species of most concern should be those with high mortality and low recruitment at most localities, and these should be the focus of research to explain the paucity of regeneration and for restorative actions. Such demographic data can then, in part, be used by DOC to prioritise tree species in terms of risk. Internationally, there is considerable interest in the maintenance of tree species populations, with 10% of all the world's tree species considered to be under threat (Williams 1998).

TABLE 3. DEMOGRAPHIC RATES OF FOUR TREE SPECIES FOR STEMS > 10 cm dbh IN FORESTS AT VARIOUS LOCALITIES IN NEW ZEALAND.

P = mean percentage each species forms of total stems per plot; M = mortality rate (%/yr); R = recruitment rate (%/yr); T = turnover rate (mean of mortality and recruitment rate, %/year). After Bellingham et al. (1999b).

LOCALITY	FOREST TYPE	SILVER BEECH				MOUNTAIN BEECH				HALL'S TOTARA				KAMAHI			
		P	M	R	T	P	M	R	T	P	M	R	T	P	M	R	T
Pirongia	Hardwood													38.2	2.4	3.1	2.8
Okataina	Hardwood-conifer													32.0	0.8	0.3	0.6
Pureora	Hardwood-conifer													6.5	4.9	1.7	3.3
Kaimanawa	Mixed <i>Nothofagus</i> -hardwood	55.7	2.6	2.5	2.6									10.1	3.6	3.5	3.6
Kaweka	<i>Nothofagus</i> (mono-dominant)					79.5	2.4	1.3	1.9	4.6	1.9	2.0	2.0				
Taranua	Mixed <i>Nothofagus</i> -hardwood	27.8	0.3	0.6	0.5					2.0	0.2	0.3	0.3	52.8	0.4	0.7	0.6
Mt Arthur	Mixed <i>Nothofagus</i>	46.8	0.3	1.1	0.7	51.4	1.0	1.2	1.1								
Kokatahi	Hardwood-conifer									2.8	3.0	0.0	1.5	24.2	0.8	0.8	0.8
Whitcombe	Hardwood-conifer									9.2	1.4	0.2	0.8	27.9	0.5	0.6	0.6
Craigieburn	<i>Nothofagus</i> (mono-dominant)					99.9	1.3	0.4	0.9								
Caples	Mixed <i>Nothofagus</i> -conifer	39.3	0.9	0.6	0.8	27.1	0.5	0.7	0.6	5.5	0.4	0.0	0.2				
Greenstone	Mixed <i>Nothofagus</i>	47.1	1.0	1.0	1.0	33.8	1.2	1.1	1.2								
Murchison	Mixed <i>Nothofagus</i> -hardwood	27.6	1.1	0.9	1.0	44.7	0.9	0.7	0.8					9.2	0.9	1.0	1.0
Waitutu	Mixed <i>Nothofagus</i> -hardwood-conifer	22.1	0.8	0.7	0.8	14.8	1.2	0.7	1.0	3.4	0.7	0.6	0.7	35.8	0.6	0.8	0.7

4.3 COMMUNITY COMPOSITION

Most approaches to species monitoring deal with a scalar quantity such as the abundance of an individual species. Species composition at monitored sites is a multivariate quantity, which may be analysed in terms of presence or absence, abundance, or proportional abundance of each species simultaneously (Philippi et al. 1998). Such multivariate data are commonly analysed using widely available classification and ordination programs (e.g. CANOCO). The CANOCO algorithm has the option of partitioning compositional variation into components explained by various explanatory variables (e.g. Økland & Eilertsen 1994). These explanatory variables could represent the pressure exerted by a specific human-related impact, such as deer density or amount of deer browse. Thus, in establishing the status of indigenous forests, it could be asked: to what degree does overall compositional variation in indigenous forests relate to levels of deer browse in a way that is unrelated to other factors (e.g. canopy density or climate) influencing composition? A logical extension of this approach is to introduce time as another factor. Wiser et al. (1997) used permanent plots to examine how the compositional recovery of mountain beech forest following fire is influenced by distance from the unburnt forest margin alone, time since the fire alone, and the interplay between time and distance. To use the deer browse analogy, we could ask: to what degree are temporal changes in indigenous forest composition related to deer browse intensity alone, deer browse intensity that is at the same time related to other factors, or to other factors alone? The challenge is to collect the long-term data on species composition, and other variables, that will allow testing of such explicit questions. Species composition can be recorded on permanent plots re-measured over time to provide the time-series data, but it must be appreciated that compositional patterns at a point in time, and compositional dynamics over time, relate to the spatial scale of sampling (e.g. Reed et al. 1993). The compositional time-series data can readily be collected for vascular plants, possibly for birds and, with more difficulty, for bryophytes, lichens, and invertebrates. There is the opportunity for further work on the extent to which one guild of organisms can represent another guild, and what would be the appropriate guild to select for monitoring (e.g. Crisp et al. 1998). Another possibility is that, within sampling design constraints, groups like bryophytes and invertebrates could be sampled on a subset of plots. To have confidence that changes in composition over time are meaningful, it is essential that consistent standards are maintained in field identification skills.

Compositional data are also needed to update the characterisation of biotic assemblages, e.g. forest types. There is a tendency to believe that forest types are fixed over time. This is clearly not the case, with some species being lost over extensive areas, for example as a result of dieback, and new mixtures being formed with exotic species. An updated classification of forest types could be the basis for subsequent subdividing of the LCDB area (see Section 4.1).

The need to measure changes in species composition also arises because some authors argue that overall composition may be the best indicator of human-related changes in vegetation. In this way, Stephens et al. (2002) used the proportion of indigenous species remaining as an index of biota removal for assessing conservation achievement. Compositional data also allows taxa to be

combined into groups that reflect specific modifying forces on indigenous biodiversity. For example, what would be the implications of a decline in the density of birds feeding on the fruits of indigenous plants? A reduction in frugivorous birds may lead to a reduction in the distribution of bird-dispersed plants. It may be appropriate to group those plant species of which the fruits are eaten by birds (e.g. Cameron et al. 1987), for such analyses. That is, the patterns may be more clearly expressed when the data are analysed at higher levels than species.

4.4 EXOTIC WEEDS

Plant communities worldwide are becoming progressively homogenised by the success of exotic species in new regions, and New Zealand is no exception to this trend. Traditionally, there has been a focus on evaluating the susceptibility and consequences of exotic weed invasions in managed or disturbed habitats. The 15% of New Zealand's original lowland forest that remains is often fragmented and is considered vulnerable to weed invasions, particularly along edges and in small remnants (Owen 1997). Naturally disturbed sites within the indigenous forest matrix are also invaded by many exotic species. Although this is commonly by herbaceous species, it is increasingly apparent that exotic woody species are playing an early successional role. For example, radiata pine (*Pinus radiata*) widely establishes on landslide surfaces in Urewera National Park, where it will remain a structural dominant for many decades, and sycamore (*Acer pseudoplatanus*) invades natural canopy gaps in some South Island forests. Exotic tree species are not confined to invading disturbed forest, Douglas fir (*Pseudotsuga menziesii*), for example, can establish under open, natural beech forest canopies in some areas (Ledgard 2002). Relatively undisturbed forests are also being invaded by plant species such as old man's beard (*Clematis vitalba*) and wild ginger (*Hedychium gardnerianum*), as well as exotic herbaceous species (Wiser 2000). The number of naturalised plant species that have become weeds in New Zealand has been steadily growing since the 1860s and, with the large pool of introduced species in the country, this trend is unlikely to slow down. A response to this by DOC is to maintain a database of weeds on conservation lands (Owen 1997).

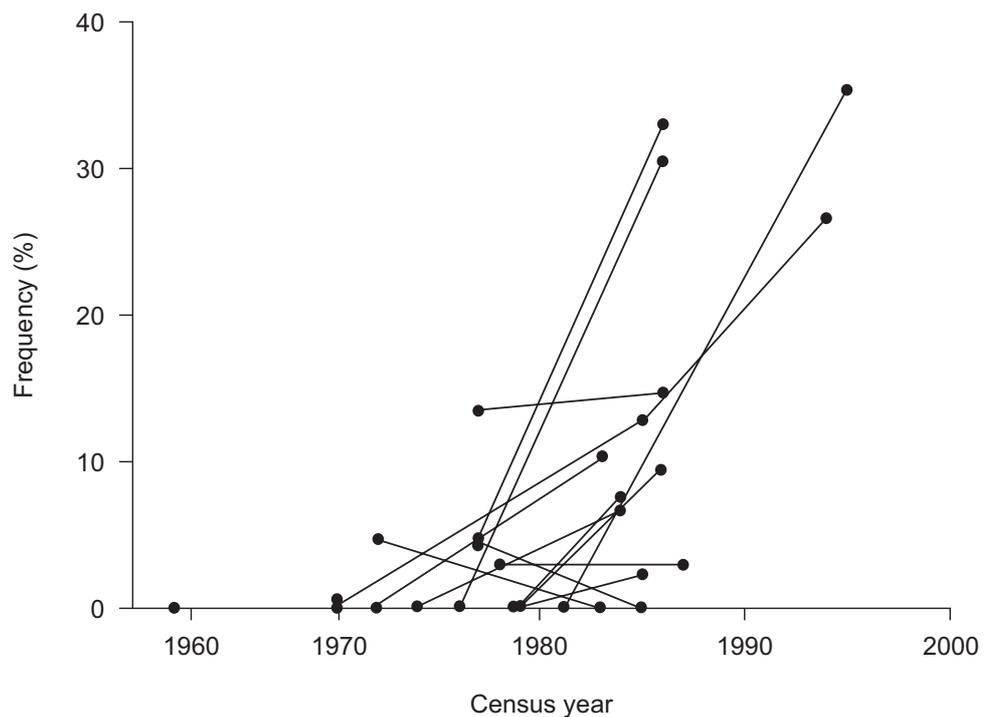
Key issues with invading species are how widely distributed will they become, and how will they influence the status of conservation assets over time. Wiser et al. (1998) have shown for *Hieracium lepidulum* invading Canterbury mountain beech forests that our ability to predict the weed's distribution increases during the invasion, but also that the site variables that relate to a weed's distribution change over time. There is also no reason to expect that the distribution of a species in its original habitat will necessarily be strongly related to its distribution elsewhere (e.g. Panetta & Mitchell 1991). For example, radiata pine grows in New Zealand in conditions very different from its native Mediterranean climate, rocky headland environment in California. It would seem plausible that species with limited distributions in their natural range may expand their habitat range in new locations when there is competitive release. A limitation of a national or regional monitoring system is that it is only likely to evaluate the more common species. More restricted

species, perhaps those species in the early stages of invasion, will need to be evaluated by other means. There has been little work done on the impacts of weed invasions in indigenous forests, yet weeds can potentially alter tree regeneration, understorey composition, litter fall, and decomposition processes.

Most early accounts of indigenous forest weed invasions are anecdotal, although herbarium records provide a means of assessing presences at some locations in the past. Weed distributions can also be assessed using plot data. Although plot-based analyses are possible for some species, e.g. *Mycelis muralis*, for many exotic herbaceous species (e.g. graminoids) the taxonomic abilities of field parties have been inadequate to confidently identify sites where exotic species are present or, just as importantly, where they are absent. This highlights the need for DOC staff to have good taxonomic skills. The network of permanent plots being established by the USA forest health assessment is using the percentage of species exotic as its indicator of exotic invasion (Stapanian et al. 1998). It would be useful to add to this the percentage cover of exotic species. Such analyses will clearly result from collecting the type of compositional data outlined in the previous subsection.

Permanent plots can certainly provide a reliable means of assessing how the distribution and abundance, as well as some impacts, of weed species change over time. For *Mycelis muralis*, we can show that time-series data from re-measured plots in South Island survey areas generally show a trend of increasing frequency of this species, although there is variability in the rate of increase (Fig. 7). The impact of such weed invasions can be assessed on permanent plots through related changes in seedling numbers or other herbaceous species.

Figure 7. Changes in the percentage frequency of *Mycelis muralis* on re-measured plots in 12 South Island survey areas (S. Wiser unpubl. data).



4.5 IMPACTS OF INTRODUCED ANIMALS

There has been pervasive modification of New Zealand's indigenous forest understoreys by introduced browsing animals. In some instances, where animals remain in high numbers, the consequence is an open forest understorey with the few remaining plants heavily browsed. Elsewhere, a dense understorey has developed of species unpalatable to the browsing animals. It is now becoming apparent that this browsing has consequences for a much wider spectrum of indigenous biodiversity and ecosystem properties than has usually been considered (Wardle et al. 2001). Some of these compositional changes will remain as a legacy of former times for well into the future. An early approach to assessing browsing impacts was repeat photo points in the forest understorey. Although this approach requires little effort, the photographs cannot provide detail on compositional changes or clearly define the level of browse. To some degree all browse assessments are found lacking in that they have not been clearly linked to the conservation benefits gained from management intervention. For biota, we need to ask: what is the significance, in demographic terms, of a certain level of browse? As can be seen in other parts of this report, the demographic information, and a measure of browse, could serve as a way of linking demographic performance to animal impacts.

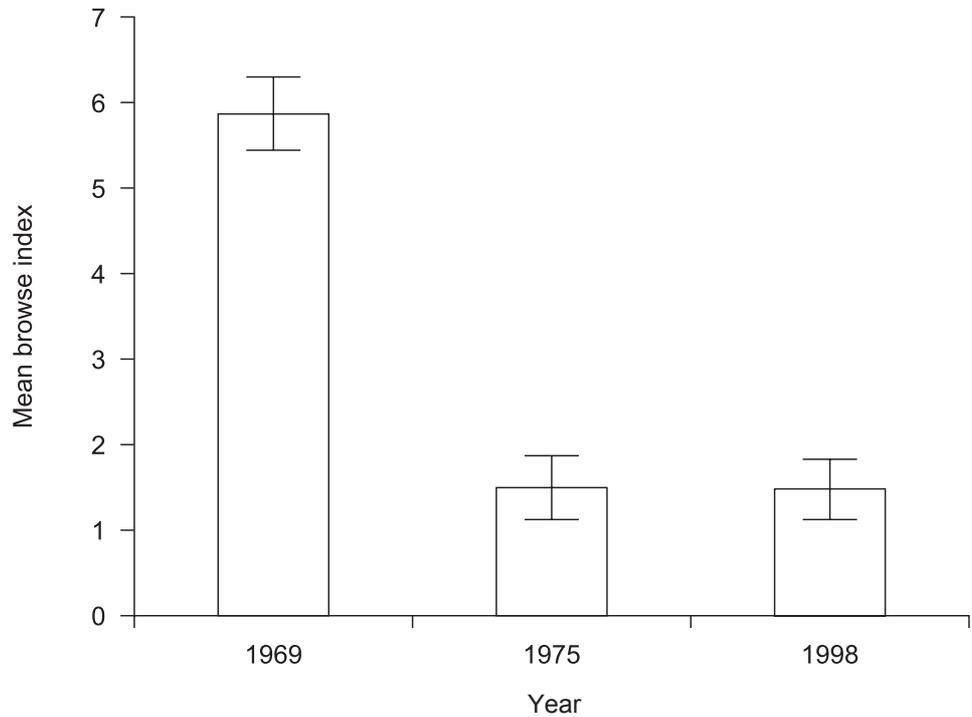
Several browse indices have been extensively tested and used over the last 30 years (Wardle et al. 1971; Rose & Burrows 1985; Payton et al. 1997). Some of the common plot-based forest understorey indices useful for ungulates are:

- Browse index An estimate of the total amount of browsing on a species over a group of plots
- Percentage total browse The amount of browsing on a species as a proportion of total browsing on all species over a group of plots
- Browse pressure index A measure of the amount of browse on a species relative to its availability
- Mean browse index A measure of the browsing intensity on all species over a group of plots

These browse indices are simply derived from quick-to-make observations made of plant species browsed on plots. Comparisons over time can be used to indicate changes in the impact of browsing animals (Fig. 8). Because of the observational nature of this monitoring, there needs to be careful standardisation of procedures, e.g. search effort, among observers over time. In addition, it can be difficult to distinguish the damage caused by various animals, e.g. goats versus deer, if this is required. Because the observations are made on foliage, any browse observed is also a short-term impact, relative to the periodicity of plot remeasurement, and can be confounded with periodicity in leaf production.

There has been a history of canopy assessment techniques being developed in New Zealand, mainly because of defoliation brought about by browsing by brushtail possums. Payton et al. (1999) list techniques including descriptive accounts, photo points, hemispherical photography, canopy scoring, aerial photography, airborne video, and satellite imagery. The advantage of direct observations of tree canopies, and associated canopy scoring, is that the link to

Figure 8. Mean browse index on plots in the Murchison Mountains over the last 30 years (Burrows et al. 1999).



possum browsing is clearer than for other methods. The Foliar Browse Index (Payton et al. 1999) is a widely used method that combines a canopy-scoring approach with the use of indicator species to provide an assessment of damage. With appropriate statistical considerations, this index can be estimated on tagged individuals, and make a long-term contribution to linking defoliation (or browsing) to impacts on demographic parameters.

A wide array of methods is used for assessing the abundance of animal pests in New Zealand (e.g. trap catch for possums, tunnel track indices for rodents and mustelids). Although animal pest abundance is somewhat distant from actual impacts, it has been used as an index of a particular pest's impact on a range of organisms. This is a pragmatic solution to the more complex monitoring of the range of impacts, but does require validation at a subset of sites of assumed relationships. Rose & Burrows (1985) have shown a strong relationship between deer density estimates and browse indices for localities around the country. One area that needs further attention is matching the spatial pattern in pest abundance within a locality to site-specific impacts. Field-based measurements for animal abundance indices therefore need to be designed in concert with their impact assessment.

4.6 QUANTITY OF DEAD WOOD

Dead wood, as standing dead trees, fallen logs, or large branches, is a major structural component of natural forest systems (Harmon & Hua 1991). In their natural state, New Zealand's indigenous forests have large volumes of dead wood relative to temperate forests elsewhere (Stewart & Allen 1998). Dead wood performs many important ecological functions: providing habitat for organisms, playing a part in energy flow and carbon and nutrient storage, and

influencing sediment transport and storage. Woody debris at various stages of decay contributes to high levels of habitat heterogeneity. For example, the threatened long-tailed bat (*Chalinolobus tuberculatus*) selects roost sites in standing dead trees based on number of cavities, trunk surface area, and canopy cover (Sedgeley & O'Donnell 1999). Saprophytic fungal communities are diverse in logs on the forest floor, even in simple mountain beech forest, and play a role in nutrient transport from logs to the soil (Clinton et al. 1999). Many studies have shown the importance of logs on the ground for the establishment, growth and survival of tree seedlings. The contribution of dead wood to forest-stand biomass carbon can be considerable and exceed that of live biomass (Stewart & Allen 1998).

Measurements that have been made of dead wood volume, mass or production in indigenous forests have been based on permanent plots where the diameters and lengths of individual pieces of dead wood have been measured (Stewart & Allen 1998). Usually the material is classified into decay classes representing a decay sequence that influences the ecological functions of dead wood. We know little about the rates at which dead wood decays, or is lost from the dead-wood pool, although Stewart & Burrows (1994) have shown that red beech (*N. fusca*) logs that are decayed by lignifying brown rot fungi take centuries to decay, whereas adjacent silver beech logs decayed by white rot fungi disappear in a few decades. Such relationships mean there is, at best, only a weak relationship between live and dead tree biomass in forests. Because of this, modeling the amounts of dead wood in forests (for example, as a reciprocal oscillation between live and dead mass) will be complex, and it will be more accurate to directly quantify the amounts of dead wood. The only available method for determining the volume of dead wood in forests, and its characteristics, is direct measurement in the field. This approach has been implemented in the carbon inventory system for indigenous forests funded by the Ministry for the Environment (Coomes et al. 2002).

4.7 POTENTIAL INDICATORS

There is a wide range of compositional, structural or functional indicators that could be used in forests that have not so far been discussed. With time, these can be tested, added to and, maybe in a few instances, replace those described so far. Some of the indicators currently being tested are briefly considered below, to show the breadth of possibilities.

4.7.1 Phylogenetic diversity measures of biodiversity

(prepared by Gary Barker, Landcare Research, Hamilton)

The IUCN (1980) interpreted the primary reason for preserving biodiversity as being the maintenance of options for the future: the greater the complement of contemporary biodiversity we are able to conserve today, the greater the possibilities for biodiversity in the future because of the diverse genetic resource needed to ensure continued evolution. From this perspective, biodiversity option value can be equated with species richness, plus the richness of activities each species undergoes during its existence through events in the life of its members, plus the non-phenotypic expression of its

genome. Resource managers needing to estimate richness of features are usually unable to measure this directly or, at best, are only able to do so for a small sample of genes or characters. However, because genes and characters are inherited, the relative feature richness and, thus, biodiversity value of different communities can be estimated using knowledge of genealogical relationships among the species that make up those communities. Phylogenetic Diversity (PD; or Feature Diversity) was first proposed by Faith (1992) as a measure of biodiversity option value. This measures how much of the pattern of feature diversity, i.e. how much of the branching pattern in the phylogenetic tree, is represented in a sample of species (see Barker 2002).

At the basic level, PD calculated from lists of species at particular sites, or regions, enables reporting on contributions to biodiversity at larger spatial scales. For example, from a list of birds found in the region, we can express the contribution of Canterbury to national and global bird diversity. When we have species inventories at different points in time, we are able to monitor trend in contribution. At this basic level, changes in contribution are brought about by extinctions, either at the population or species level (depending on the scale of observation). Appropriate software can utilise abundance data in the derivation of PD scores, thus enabling the detection and monitoring of changes in community structure in the absence of population/species extinctions. Thus, PD scores can be derived for communities that are sampled repeatedly over time. Such an index therefore can make use of the community composition data described in Section 4.3.

4.7.2 Litter quality

(prepared by David Wardle, Landcare Research, Lincoln)

Generally, our studies on human-related changes in indigenous forests have focussed on above-ground components. This would seem appropriate given, for example, the pervasive changes in forest understoreys caused by deer browsing (e.g. Ogden et al. 1996). However, the impacts are more complex—plant material enters the decomposer subsystem as litter where its breakdown is partially controlled by substrate quality (Wardle et al. 1997). Browsing animals are expected to preferentially eat foliage of species whose litter decomposes most rapidly, since the controls on palatability and decomposition of foliage are similar (Wardle et al. 2002). The subsequent increase in species with unpalatable foliage will have important consequences for below-ground processes. In a more general sense, any human-related changes in litter type and composition (e.g. exotic plant species) will have flow-on effects for how forest ecosystems function. These flow-on effects drive the vigour of forests and successional change. Slow decomposition can be linked to carbon accumulation in soils and low fertility with, as a consequence, plants growing more slowly and considerable change in forest composition. The leaves of plants that store nitrogen in secondary metabolites will not readily decompose and nitrogen will be unavailable and will accumulate in litter and soils, rather than being taken up by plants. Current research is identifying how human-related impacts modify litter quantity and quality in indigenous forests as well as the consequences of these changes. It is expected that indicators for assessing such changes in forest litter quality will become available and provide an important link between above- and below-ground impacts.

4.7.3 Forest vigour

(Prepared by Neal Scott, Landcare Research, Palmerston North)

Measurements of forest productivity (and changes through time) represent an integrated measure of forest vigour. Net primary production (NPP) is a measure of the net growth of a forest over some period of time and, in its simplest form, is represented by the equation:

$$\text{NPP} = \text{GPP} - \text{RESP} - \text{MORTALITY}$$

where GPP is gross primary production (the amount of carbon dioxide absorbed by the canopy), RESP is stand-level respiration from all vegetative components, and mortality is death of part or all of a tree. Stand-level NPP follows a general pattern during stand development for even-aged stands: it increases rapidly during the early stages of stand development, and peaks during the early to middle stage of stand development, after which it declines in older stands (Landsberg & Gower 1997). The cause of this decline is not clear, but current evidence suggests that hydraulic limitation of photosynthesis is one mechanism (Murty et al. 1996). In mixed-age-class forests, a quasi-steady-state NPP may be achieved as small gaps arising from tree mortality or other disturbances (e.g. herbivory) are filled with smaller, more rapidly growing younger trees. As forests are disturbed, their NPP may change as the stand is 'reset' from a mature, low-productivity forest to an actively growing younger stand. Information on NPP can indicate just where on the continuum of productivity, or vigour, a forest resides and how resistant it will be to certain stresses.

Several factors interact to regulate NPP, including the amount of leaf area and its chemical composition, species composition (which influences foliar and physiological characteristics), nutrient availability, water availability, humidity, and solar radiation. A forest may be viewed as a large green sponge that absorbs light (energy), and uses water and nutrients to produce biomass. Given information on the quantity and quality of canopy leaf area (that influences light absorption and utilisation), and general information on soil fertility and climate (rainfall and temperature), forest NPP can be estimated with reasonable (and verifiable) accuracy. Climate and soils data, plus forest physiological data, can be used to predict the maximum achievable NPP for a given site. If measurements indicate significant deviations from the predicted values, then we have an indication of sub-optimal vigour related to (non-climatic) disturbance. Given that climate and soil fertility do not change rapidly, it is information on canopy condition that determines variation in NPP over shorter time scales.

How can information on forest canopies be obtained? At small spatial scales, the amount and 'quality' of leaf area can be measured directly. At larger spatial scales, remote-sensing techniques can provide measures of leaf area and chemical composition that are sufficiently accurate to predict NPP over large areas (e.g. all of New Zealand). This information can then be used in conjunction with climate and soils information to assess NPP for all New Zealand's indigenous forests. What this provides is a broad-scale, spatially integrated estimate of forest vigour that can be updated frequently. Future satellite-based sensors are likely to improve our ability to monitor forest NPP (and other properties), as new sensors are being designed specifically to measure critical forest and forest canopy parameters.

4.7.4 Phenological events

A feature of the New Zealand flora is periodic heavy flowering and fruiting in plants. There are well-established links between phenological events and predator populations, which can have important conservation implications for invertebrate and threatened bird populations. The long-term studies in the Orongorongo Valley, near Wellington, and at other localities, bear testament to these links. The breeding success of threatened bird species may also be linked to phenological events, so that periodic breeding in kakapo appears to have been associated with heavy flowering and fruiting in native plant species, especially podocarps (Lee et al. 1997). If such relationships do exist, then determining which factors are correlated with the initiation and development of flowering and seeding events in indigenous plants may provide early indications of a need for conservation management intervention. Higher seed production in many indigenous species exhibiting periodic heavy seeding has been correlated with higher summer temperatures at the time of floral primordia initiation, approximately 1 year before flowering. This climatic relationship has been quantitatively shown for the *Nothofagus* species, at a range of localities, and hinau (*Elaeocarpus dentatus*) (Lee et al. 1997); indicating that there can be high levels of intra- and inter-generic synchrony over large spatial scales (Schauber et al. 2002). Rimu (*Dacrydium cupressinum*) is an exception to these relationships; its seeds fall 2 years after floral initiation and the amount of seed produced is negatively correlated with summer temperatures at this time. Because temperatures in New Zealand may vary with the El Niño-Southern Oscillation, it may be that subsequent seeding periodicity across genera is controlled by these climatic events. Schauber et al. (2002) have shown that the intra- and inter-generic synchrony of seeding events throughout a large part of New Zealand partly result from a shared climatic response linked to the Southern Oscillation.

As our ability to predict the occurrence of periodic heavy seeding improves, so too will our ability to initiate a timely intervention response to minimise repercussions for conservation assets. In this instance, the indicator may well not be the status of human-related impacts (e.g. levels of predation by stoats), nor the pressure on the assets (e.g. the number of stoats), but an indicator that will subsequently determine the pressure and state. Ultimately, it may mean linking ocean temperatures and stoat impacts. This is a desirable state because it will allow DOC to be predictive and, hence, through appropriate resource allocation, increase conservation performance.

4.8 INDICATORS AND CONSERVATION PERFORMANCE

The indicators outlined in this report are relevant to assessing conservation management. Certain indicators can stand alone. For example, increasing cover or proportion of exotic species reflects the status of the conservation asset base, and both are ordinal and reflect a decline in status. Certainly, quantitative data on such indicators over time are considerably more robust than anecdotal impressions. Various indicators can also be combined that reflect status of indigenous biodiversity with and without management. Tree species that may

be of most concern would be those that have mortality exceeding recruitment rates over a large proportion, rather than just a small part, of their distributional range. Such demographic data can be used not only to assess the status of tree species, but to also make predictions about, for example, future extinction dates. The imbalance between tree mortality and recruitment rates could then be combined with the levels of exotic weeds to provide a more integrative index of the status of conservation assets. The reasons, approaches and merits for combining indicators into other 'synthetic' indicators still require a considerable amount of work.

5. Links with other initiatives

5.1 NATIONALLY

There is a large degree of commonality among indicators proposed in this report for forests and those proposed for or used in various other initiatives for assessing a wider range of ecosystems (Table 4). The terminology used does vary and there may be gains from using a common terminology, where this is possible. The level of commonality is not surprising when one considers the limited range of attributes that can be measured in these ecosystems that reflect our concerns. That each, in part, uses the LCDB highlights the importance of this land use/cover map being robustly designed. Below we briefly describe any differences between indicators proposed in this report and those used in various other initiatives (detail shown in Table 4). The indicators and methods proposed in this report are the most detailed for forests.

5.1.1 Environmental Performance Indicators

The Environmental Performance Indicators (Ministry for the Environment 1998) require cadastral information not described in this report and also include a range of indicators beyond the scope of this report (e.g. Bio9, Bio10, Bio13; Table 4).

5.1.2 Conservation Achievement

The indicators proposed for Conservation Achievement (Stephens et al. 2002) have been designed for all conservation lands. It is not surprising that they are very similar to those proposed in this report for assessing the status of forest conservation assets. The only real difference is in the indicator of resource modification proposed by Stephens et al. (2002). This is potentially very useful but has not been included in this report as further research is needed to define appropriate indicators for its use in forests—some options were addressed in Section 4.7.

5.1.3 Carbon Monitoring System

The data required for the Carbon Monitoring System (Coomes et al. 2002) can be obtained from a subset of the indicators proposed in this report.

TABLE 4. COMPARISON OF FOREST BIODIVERSITY INDICATORS PROPOSED IN THIS REPORT WITH THOSE CONTAINED IN OTHER RELEVANT NATIONWIDE INITIATIVES.

The level to which data required for indicators in this report contributes to indicators in other initiatives is described.

THIS REPORT	ENVIRONMENTAL PERFORMANCE INDICATORS (update of Ministry for the Environment 1998)	CONSERVATION ACHIEVEMENT (Stephens et al. 2002)	CARBON MONITORING SYSTEM (Coomes et al. 2002)
<p>1. Forest area: LCDB would be used to derive changes in indigenous forest cover and geometry over time. Plot data would allow definition of further forest cover types, and their changes, based on updated actual data.</p>	<p>Bio1 and Bio2 (and Bio4): land use and cover also derived from LCDB. Further refinement of land cover could come from geo-referenced plot data. Bio3: plot data would characterise areas (ecosystems, habitats) Bio5: forest fragmentation indices would also come from changes in geometry over time.</p>	<p>Fragmentation: spatial relationships also derived from LCDB and expressed as a fragmentation index. Measures include isolation, edge exposure, and size.</p>	<p>Total forest area from the LCDB is required for estimating biomass carbon storage.</p>
<p>2. Tree mortality and recruitment: the maintenance of structural dominance will be determined from plot-based data.</p>	<p>Bio6: plot data provides a condition measure based on the maintenance of community structure.</p>	<p>Biotic removal: plot-based data would provide a structured field measurement of removal of structural dominants.</p>	<p>Individual tagged tree data is used to calculate tree- and stand-level biomass carbon storage.</p>
<p>3. Community composition: the maintenance of community assemblages would be based upon plot data</p>	<p>Bio6: plot data provides condition measures based on abundance, community composition/properties. Bio12: plot data could provide measures of evolutionary diversity in some taxonomic groups (could include birds).</p>	<p>Biotic removal: the plot-based measurements could provide comprehensive measurements of human-induced removals of the native biota at geo-referenced points on the landscape.</p>	<p>Vascular and non-vascular plant composition recorded on plots for carbon project—could be expanded to other types of organisms.</p>
<p>4. Exotic weeds: plot data would provide data on the distribution, cover and abundance of more common weeds.</p>	<p>Bio8: cover and abundance of common weeds would be determined over time on plots.</p>	<p>Competition pressure: plot-based data provides cover of various common weed species and, importantly, with re-measurement, the displacement of native biota.</p>	<p>Plot data provides exotic weeds data used in carbon project.</p>
<p>5. Browsing impacts of introduced animals: actual impacts of selected pests recorded and abundance of pests.</p>	<p>Bio6: plot-based browse recording gives an index of modification by herbivore pests. Bio7: browse indices based on plot data can be strongly related to herbivore abundance.</p>	<p>Consumption pressure: the plot data collects information on the impact of vertebrates on plants and indices of pest abundance (e.g. predators).</p>	<p>Browse indices derived from plot-level observations are incorporated in carbon monitoring system.</p>
<p>6. Quantity of dead wood: amount, and stage of decay, measured on plots and used as a measure of community structure.</p>	<p>Bio6: plot-based measurements give an index of woody debris as a community structure measure of habitat.</p>	<p>Biotic removal: modification of woody debris regimes can indicate human-induced modification of native biota.</p>	<p>Dead wood biomass carbon storage is based on plot-level measurements of dead wood volume, decay stage and mass.</p>

5.1.4 Biodiversity Strategy

The common set of indicators among the initiatives outlined in this report have the potential to contribute to many of the Action Plan objectives listed in theme 9 of the New Zealand Biodiversity Strategy (Department of Conservation and Ministry for the Environment 2000). Although the types of information required for the Action Plan objectives are not very explicit, they illustrate the links between these objectives and this report:

- Objective 9.2—develop and implement effective approaches to map indigenous biodiversity at ecosystem scales and inform management actions and research.
- Objective 9.3—use consistent measures and methods to monitor and provide information on key changes in the extent and condition of indigenous biodiversity.
- Objective 9.4—ensure that local, regional and national reporting on the state of indigenous biodiversity informs ongoing priority setting for biodiversity management and research as a key part of an adaptive management approach.

As a consequence, the methods and indicators proposed in this report for forests, and shared with other initiatives, would contribute in a major way to the Biodiversity Strategy. This would mean that there would not be a need to redesign methods and indicators for the Biodiversity Strategy.

5.2 INTERNATIONALLY

A variety of international environmental conventions and agreements require New Zealand to report on key environmental criteria and indicators. Bellingham et al. (2000) list initiatives with reporting requirements—although few of these have, as yet, defined criteria and indicators. New Zealand has already reported under the Montreal Process (Ministry of Forestry 1997) and several issues were outlined that constrain New Zealand's ability to adequately report on its forests. These were:

- Information not being available
- Information being available but considerable work required to collate it
- Custodians charging for access to information

There is a need for a consortium approach to reporting on forests, and other ecosystems, and this approach needs to be predicated on an understanding of the full range of needs both nationally and internationally. International agreements for which there are developed criteria and indicators, as well as an agency responsible for administering the agreement, include:

- Convention on Biological Diversity—Department of Conservation and the Ministry for the Environment (MfE)
- Forest Resource Assessment 2000—Ministry of Agriculture and Forestry (MAF)
- Montreal Process—MAF
- Land use and forest changes under the Intergovernmental Panel on Climate Change—MfE

In the following subsections the extent to which the indicators described in this report for national reporting satisfy these international agreements is briefly considered.

5.2.1 Convention on Biological Diversity (CBD)—indicators of forest biodiversity

The CBD focuses on the conservation of biological diversity and its sustainable use—this is reflected in the indicators proposed by a CBD liaison group on forest biological diversity (see Appendix 1). The International and Core Sets of indicators proposed for forests are usually met by the indicators described in this report. In fact, this report develops—much more than the CBD documentation—what can be measured for the ‘forest condition’ indicator. Additional mapping information (that is largely available e.g. ‘protected areas by ecoregions’), in combination with indicators in this report, would allow virtually all of the International and Core Set indicators to be reported on. When the indicators proposed in this report are combined with (generally available) mapped information as well as classifications of species (e.g. levels of threat), virtually all of the CBD Detailed National indicators (Appendix 1) can be reported on. The remaining indicators not covered largely relate to sustainable use—information that should be available from MAF.

5.2.2 Forest Resource Assessment 2000 (FRA 2000)

The FRA 2000 indicators focus on forest area, biomass and carbon stocks. As a consequence these indicators consistently use the forest area and tree information indicators in this report (see Appendix 2). Again, by adding available mapped (e.g. cadastral) and other (e.g. soil carbon from Carbon Monitoring System) information in combination with the indicators in this report, most of the core indicators are available. Some of the indicators that should be ‘attempted’ have been used in New Zealand (forests managed primarily for soil protection) and could be obtained from mapped information.

5.2.3 Montreal Process

The indicators proposed in this report address at least some of the indicators in Criteria 1-5, but not Criteria 6 and 7, of the Montreal Process (see Appendix 3). For Criterion 1 (Conservation of Biological Diversity), all the indicators are met when the indicators in this report are combined with available mapped information and a classification of species (e.g. level of threat). Criterion 2 is about production from forests, and the background resource information is provided by the indicators in this report but not the information on levels of production (this is mostly available from MAF). The indicators in this report cover Criterion 3 (Forest Health and Vitality)—except for those indicators related to pollution. In New Zealand, pollution assessment in trees is currently restricted to the immediate vicinity of major processing plants. Criterion 4 is concerned with soils and water resources; these are not covered in this report except to the extent that forest area data provide some basis for erosion assessment. The indicators in this report largely meet the carbon reporting requirements of the Montreal Process (Criterion 5), but not the indicators on forest products and carbon. Criteria 6 and 7 deal with socio-economic and legal issues not covered in this report.

5.2.4 **Land use and forest changes under the Intergovernmental Panel on Climate Change (IPCC)**

The first three IPCC indicators on land use and forest changes are covered by the indicators in this report as they relate to forest biomass and area changes (Appendix 4). The other two indicators on CO₂ emissions are covered by the Carbon Monitoring System and other emissions work.

5.2.5 **Summary**

Those agencies responsible for reporting under these international agreements need to carry out a similar exercise to that which is shown in the appendices of this report—comparing what is required by the convention with what monitoring data is already available in New Zealand. Only then can we comprehend how completely New Zealand can report on these agreements. One issue which makes uniformity of approach between New Zealand and international agreements difficult is the type of information commonly collected in New Zealand. At present there is much emphasis on biodiversity, and how pests and weeds affect this, and little emphasis on soil and water values. Ironically, 50 years ago abiotic values (such as soil and water) were the principal values for which much of New Zealand's indigenous forests were managed. This change illustrates that values we monitor for today may not be those of the future. It is interesting to note that the significance of soils is again being recognised in some recent work (e.g. Wardle et al. 2002).

6. Conclusion

The following indicators could be used to monitor forests in a systematic fashion over all land in New Zealand:

- Forest area as a habitat indicator
- Mortality and recruitment rates of trees for maintenance of structural dominants
- Community composition as an indicator of species assemblages
- Exotic weeds as a measure of intactness of indigenous biodiversity
- Indices for introduced animal impacts
- Quantity and characteristics of dead wood as one indicator of habitat diversity

Many of these indicators are currently best assessed through a network of permanent plots, but this could change with the development and testing of new technologies (e.g. satellite-based observations). Long-term observations on permanent sample plots is, however, the only approach that provides real data on patterns and rates of change in forest ecosystems, and is the ultimate test of other approaches (Acker et al. 1998). There is also considerable merit in having indicators which can be used in predictive models to develop time-frames for management intervention.

This report complements Bellingham et al. (2000) who review plots as long-term monitoring sites and Allen et al. (2002) who discuss how to use such data to prioritise concerns about the maintenance of tree species in indigenous forests.

With these reports as a basis, and in combination with the other national initiatives previously discussed, the following issues need to be addressed:

- A consortium of interested agencies and users should be established to ensure that the set of indicators chosen, in combination with other available information, forms a system that meets as many monitoring needs as possible.
- The support of senior managers from the range of agencies involved must be obtained and appropriate resources made available.
- the indicators chosen must gain wide acceptance from people involved on the operational side of forest management and must also meet international standards and expectations.

With respect to actually implementing a forest biodiversity monitoring system, the following aspects still need to be addressed:

- Certain design features of the system must be refined
- A funding stream and infrastructure for the system must be developed
- The monitoring system must then be planned and implemented
- Once implemented, the system must be reviewed and further refined according to the outcome of the reviews

Although this report was commissioned for forests, such a system should be established to cover the full range of ecosystems. Without spatially extensive and robustly designed monitoring systems, New Zealand will remain in a weak position to report on the effectiveness of biodiversity management.

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APPENDIX 1

Convention on Biological Diversity (CBD)—Indicators of forest biodiversity

<i>CBD Indicators—International</i>	<i>Current report*</i>
Forest cover	1
Forest condition	2,3,4,5,6
Protected areas	1—Partial
 <i>CBD Indicators—core set</i>	
Area of natural forest	1
Area of natural forest as a proportional of total forest	1
Change in natural forest over 10 years	1
Forests protected areas by IUCN classes I–VI	1—Partial
Forest protected areas by eco-region	1,3—Partial
Number of forest-dependent species	3—Partial
Proportion of forest-dependent species at risk	2,3
Areas of forest managed to prioritise biodiversity conservation	1—Partial
Air pollution levels exceeding forest critical loads	Not covered
Existence of legislation to protect biodiversity	Not covered
 <i>CBD Indicators—detailed national information</i>	
Mapped details of forest types	1,2,3—Partial
Mapped details of old-growth/natural forests by types	1,2—Partial
Mapped details of forest protected areas	1—Partial
Mapped details of forest under special management regime	1—Partial
Percentage and extent in area of forest types relative to historical condition and to total forest cover	1,2,3—Partial
Percentage and extent in area of forest types by age class	N/A
Levels of fragmentation and connectiveness	1
Percentage of mixed stands	1,2
Area and representativeness of forest protected areas	1,2,3—Partial
Number of forest-dependent species, categorised as (i) indigenous (ii) non-indigenous (iii) endemic	3—Partial
Number of forest-dependent species, categorised as (i) threatened (ii) endangered (iii) rare (iv) vulnerable	3—Partial
Population levels and changes over time of selected indicator species	2,3
Number of forest-dependent species, occupying a small proportion of their former range	3—Partial
Areas of forest cleared annually containing endemic species	1,3—Partial
Percentage of annual natural regeneration	2
Natural regeneration as a percentage of total regeneration	N/A
Percentage of stands managed for genetic resource conservation	Not covered
Amount of <i>ex situ</i> genetic resource conservation	Not covered
Proportion of trees suffering damage	2,5
Area of land set aside into special management regimes	Not covered
Area of land independently certified as being managed sustainably	Not covered
Human disturbance index	1,2
Main threats to forest biodiversity	2,3,4,5
Area of forest annually affected by major threats	1,2,3,4,5

***Indicators in this report cover:**

1 = Forest area	2 = Tree mortality and recruitment
3 = Community composition	4 = Exotic weeds
5 = Browsing impacts of introduced animals	6 = Quantity of dead wood
Partial = Indicators in this report only partially cover CBD 2000 indicator	
Not covered = CBD indicator not covered in this report	
N/A = Not relevant to indigenous forests	

APPENDIX 2

Forest Resource Assessment 2000 (FRA 2000)

<i>FRA 2000—Core indicators</i>	<i>Current report*</i>
Area of forest	1
Areas of other wooded land	1
Area of forest by naturalness	1,2,3,4,5,6
Area of forest plantations by categories of species	N/A
Forest areas converted to other uses	1
Total forest biomass above ground	1,2,6
Total carbon stock in forests	1,2,6
Total volume of growing stock	1,2
Changes over time of total volume of growing stock	1,2
Changes over time of total forest biomass	1,2
Changes over time of total carbon stock	1,2—Partial
Area of forest and other wooded land available for wood production	1—Partial
Area of forest by ownership	1—Partial
Area of forest in protected areas	1—Partial
Area of forest and other wood land burned annually	Not covered
Biomass of forest types (broadleaf and coniferous)	1,2,6
Quantity and/or total value of harvested non-wood goods and services	Not covered
<i>FRA 2000—Indicators for which assessment should be ‘attempted’ or partially made</i>	
Fragmentation of forests	1
Area of forest and other wooded lands managed primarily for soil protection	1—Partial
Change in defoliation over past 5 years (if not FRA 2000, then later)	1,5
Area of forest and other wooded lands managed primarily for tourism and amenity	1—Partial
Area of forest and other wooded lands managed primarily for water protection	1—Partial
Maintenance of cultural, social and spiritual values	1—Partial

***Indicators in this report cover:**

1 = Forest area	2 = Tree mortality and recruitment
3 = Community composition	4 = Exotic weeds
5 = Browsing impacts of introduced animals	6 = Quantity of dead wood
Partial = Indicators in this report only partially cover FRA 2000 indicator	
Not covered = FRA 2000 indicator not covered in this report	
N/A = Not relevant to indigenous forests	

APPENDIX 3

Montreal Process

Criterion 1: Conservation of biological diversity ***Current report****

Ecosystem diversity

Extent of area by forest type relative to total forest area	1,2,3
Extent of area by forest type and by age class or successional stage	1,2,3
Extent of area by forest type in protected area categories as defined by IUCN or other classification systems	1,2,3—Partial
Extent of areas by forest type in protected areas defined by age class or successional stage	1,2,3
Fragmentation of forest types	1

Species diversity

The number of forest-dependent species	2,3—Partial
The status (rare, threatened, endangered, or extinct) of forest-dependent species at risk of not maintaining viable breeding populations, as determined by legislation or scientific assessment	2,3—Partial

Genetic Diversity

Number of forest-dependent species that occupy a small portion of their former range	2,3
Population levels of representative species from diverse habitats monitored across their range	2,3

Criterion 2: Maintenance of productive capacity of forest ecosystem

Area of forest land and net area of forest land available for timber production	1—Partial
Total growing stock of both merchantable and non-merchantable tree species on forest land available for timber production	1,2
The area and growing stock of plantations of native and exotic species	Not covered
Annual removal of wood products compared to the volume determined to be sustainable	Not covered
Annual removal of non-timber forest products (e.g. fur bearers, berries, mushrooms, game), compared with the level determined to be sustainable	Not covered

Criterion 3: Maintenance of forest ecosystem health and vitality

Area and percent of forest affected by processes or agents beyond the range of historical variation, e.g. by insects, disease, competition from exotic species, fire, storm, land clearance, permanent flooding, salinisation, and domestic animals	1,2,3,4,5,6
Area and percent of forest land subjected to levels of specific air pollutants (e.g. sulphates, nitrate, ozone) or ultraviolet B that may cause negative impacts on the forest ecosystem	Not covered

***Indicators in this report cover:**

1 = Forest area	2 = Tree mortality and recruitment
3 = Community composition	4 = Exotic weeds
5 = Browsing impacts of introduced animals	6 = Quantity of dead wood

Partial = Indicators in this report only partially cover Montreal Process indicator
 Not covered = Montreal Process indicator not covered in this report
 N/A = Not relevant to indigenous forests

Criterion 3 (continued)**current report***

Area and percent of forest land with diminished biological components indicative of changes in fundamental ecological processes (e.g. soil, nutrient cycling, seed dispersal, pollination) and/or ecological continuity (monitoring of functionally important species such as nematodes, arboreal epiphytes, beetles, fungi, wasps, etc.) 1,2,3,4,5,6

Criterion 4: Conservation and maintenance of soil and water resources

Area and percent of forest land with significant soil erosion 1
 Area and percent of forest land managed primarily for protective functions. e.g. watersheds, flood protection, avalanche protection, riparian zones Not covered
 Percent of stream kilometres in forested catchments in which stream flow and timing has significantly deviated from the historical range of variation Not covered
 Area and percent of forest land with significantly diminished soil organic matter and/or changes in other soil chemical properties Not covered
 Area and percent of forest land with significant compaction or change in soil physical properties resulting from human activities Not covered
 Percent of water bodies in forest areas (e.g. stream kilometres, lake hectares) with significant variance of biological diversity from the historical range of variability Not covered
 Percent of water bodies in forest areas (e.g. stream kilometres, lake hectares) with significant variation from the historical range of variability in pH, dissolved oxygen, levels of chemicals (electrical conductivity), sedimentation or temperature change Not covered
 Area and percent of forest land experiencing an accumulation of persistent toxic substances Not covered

Criterion 5: Maintenance of forest contribution to global carbon cycles

Total forest ecosystem biomass and carbon pool, and if appropriate, by forest type, age class, and successional stages 1,2,6
 Contribution of forest ecosystems to the total global carbon budget, including absorption and release of carbon (standing biomass, coarse woody debris, peat and soil carbon); 1,2,6
 Contribution of forest products to the global carbon budget. Not covered

Criterion 6: Maintenance and enhancement of long-term multiple socio-economic benefits to meet the needs of societies

Not covered

Criterion 7: Legal, institutional and economic framework for forest conservation and sustainable management

Not covered

***Indicators in this report cover:**

1 = Forest area
 2 = Tree mortality and recruitment
 3 = Community composition
 4 = Exotic weeds
 5 = Browsing impacts of introduced animals
 6 = Quantity of dead wood
 Partial = Indicators in this report only partially cover Montreal Process indicator
 Not covered = Montreal Process indicator not covered in this report
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APPENDIX 4

Land use and forest changes under the Intergovernmental Panel on Climate Change (IPCC)

<i>IPCC land use and forest changes</i>	<i>Current report*</i>
Changes in forest and other woody biomass stocks	1,2,6
Forest and grassland conversion	1,2,3
Abandonment of managed lands	1,2,3
CO ₂ emissions and removal from soil	Not covered
Other—emissions and removals	Not covered

***Indicators in this report cover:**

- | | |
|--|------------------------------------|
| 1 = Forest area | 2 = Tree mortality and recruitment |
| 4 = Exotic weeds | 3 = Community composition |
| 5 = Browsing impacts of introduced animals | 6 = Quantity of dead wood |
- Partial = Indicators in this report only partially cover IPCC indicator
Not covered = IPCC indicator not covered in this report
N/A = Not relevant to indigenous forests