FORUM ARTICLE

Developing a forest biodiversity monitoring approach for New Zealand

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Abstract: There is a lack of comprehensive and consistent information to inform policy makers about the status of New Zealand's forest biodiversity. Three reasons for collecting such information are: assessing the effectiveness of management, reporting on the status of biodiversity under national and international requirements, and improving our knowledge of ecosystem dynamics for designing effective management systems. The challenge is to design monitoring systems that address these multiple needs simultaneously, and at a range of spatial and temporal scales. This article first considers principles for designing enduring monitoring systems based on past experiences, assessing how effectively these principles were implemented in designing New Zealand's Carbon Monitoring System (CMS), and finally, suggesting future directions for forest biodiversity monitoring in New Zealand. At a national scale we support an unbiased, systematic sample of forests as implemented in several countries (e.g. Austria and the U.S.A.). We consider it best practice to monitor shifts in the fundamental compositional, structural and functional characteristics of ecosystems and use these to derive indicators. We suggest forest biodiversity indicators should include forest area and spatial arrangement, tree mortality and recruitment, exotic weeds, introduced herbivore impacts, and woody debris. Principles discussed in this paper are relevant to biodiversity monitoring in a wider range of ecosystems than forests. Without spatially extensive, robustly designed, biodiversity monitoring systems, New Zealand will remain in a relatively weak position nationally, and internationally, to report on the effectiveness of biodiversity conservation.

Keywords: biodiversity, carbon; forest; indicators; monitoring; national forest inventory; New Zealand.

Why monitor forest biodiversity?

The expanding influence of humans is having wide-ranging detrimental effects on biodiversity (e.g. Vitousek *et al.*, 1997). Some effects are local or regional, for example, those brought about by weed or pest introductions, while others are globally pervasive, for example, those brought about by an increasing atmospheric CO_2 concentration. However, most effects in natural ecosystems are poorly understood, in part, because we do not have systems in place to measure and analyse changes in biodiversity. The New Zealand Biodiversity Strategy presented a generalised declining trajectory for New Zealand's biodiversity (Department of Conservation and Ministry for the Environment, 2000). Although politicians and policy makers endorsed this document, it is unclear what biodiversity components were included in this trajectory and how the rate of decline was calculated. It is only known with any confidence that there has been a decline in a small proportion of the total biota (e.g. endemic land vertebrates, Atkinson and Cameron, 1993). About most taxa the evidence is anecdotal at best. Even for iconic taxa such as birds there is no large-scale, systematic measures of distribution and abundance. Kneebone *et al.* (2000) pointed out the "lack of comprehensive or consistent information that can inform policy makers about the plight of New Zealand's biodiversity". Yet a wide range of conservation expenditure in New Zealand is justified on perceived threats to, and negative consequences for, indigenous biodiversity. Some ecologists have long extolled the scientific benefits from environmental monitoring (e.g. Stewart *et al.*, 1989; Dickinson *et al.*, 1992; Wiser and Rose, 1997). However, the willingness to establish and maintain such monitoring systems has been, at best, lukewarm. Perhaps this is because the very concept of maintaining monitoring systems is anachronistic in a world where "change" is thought synonymous with "progress" (Bellingham *et al.*, 2000).

There are several reasons why New Zealand should implement a comprehensive system for monitoring forest biodiversity. In the first instance, the need to manage forests for the protection of biodiversity is enshrined in New Zealand's legislation (e.g., Forests Act 1949, Conservation Act 1987, Resource Management Act 1991). Managers of both public and private forests require information to evaluate the effectiveness of expenditure intended to protect or enhance indigenous biodiversity, especially as attempts to restore forests proliferate (e.g., Department of Conservation, 2001; Ministry of Agriculture and Forestry, 2002). Often such information comes from monitoring selected areas to evaluate specific management activities. For example, the biodiversity benefits from culling introduced animals are often assessed by comparing ecological data among areas with different treatments (e.g., Stewart and Burrows, 1989; Smale *et al.*, 1995). However, the goals for monitoring are much wider than those required to assess specific management activities (e.g., Hellawell 1991). Initiatives, both national (e.g., Ministry for the Environment's "Environmental Performance Indicators Programme") and international (e.g., Convention on Biological Diversity), require New Zealand to report on the status of its biodiversity (see Bellingham *et al.*, 2000). This is particularly necessary for forests because New Zealand is a signatory to several international agreements with specific reporting commitments for forests (e.g., Montreal Process, Food and Agricultural Organisation Forest Resource Assessment 2000). Forests are a focus of several international agreements because of their multiple roles such as carbon storage, habitat for biodiversity, ensuring water quality and timber production. Finally, monitoring changes in biodiversity will also improve our knowledge of ecosystem dynamics and thus provide the knowledge necessary to design effective forest management systems (e.g., Norton, 1996; Bakker *et al.*, 1996).

The challenge is to design monitoring systems that address these multiple needs simultaneously, and at a range of spatial and temporal scales. This article discusses some developments in forest biodiversity monitoring in New Zealand and contrasts these results with systems being developed elsewhere. We first provide principles for designing enduring monitoring systems, assess how effectively these principles were used in designing a part of New Zealand's Carbon Monitoring System, and finally suggest directions for forest biodiversity monitoring that embody these principles. Although we focus on long-term (at least decades) monitoring in indigenous forests at regional and national scales, the principles we discuss often apply to a wider range of temporal scales and ecosystems.

Principles for forest biodiversity monitoring

Based on experience in New Zealand, and the wider international literature (e.g., Noss, 1990; Dallmeier and Comiskey, 1998; Bennum, 2001), we suggest nine principles for designing a forest biodiversity monitoring system. We adopt the following definition of monitoring: It is where intermittent (regular or irregular) surveys are carried out in order to ascertain the extent of compliance with a standard or the deviation from an expected norm (Hellawell, 1991).

1. Define the problem and goals of monitoring

It is well established that the design of a monitoring system should be based upon an explicit statement of objectives (e.g., Hellawell, 1991). When long-term monitoring has the breadth of goals outlined above, it is best to specify objectives that require measurement of fundamental characteristics of forest biodiversity. The three fundamental characteristics of ecosystems are composition, structure and function (e.g., Noss, 1990; Larsson, 2001). Composition addresses the richness and diversity of species; structure, the physical organization; and function, the ecological and evolutionary processes (Noss, 1990). At any point in time these characteristics can be used to derive indicators. Because indicators, and the way they are derived and combined, continue to change with time, it is essential that long-term monitoring emphasises measurement of an enduring set of fundamental characteristics (e.g., Overton and Stehman, 1996).

2. Do not focus only on current perceptions

Although it appears efficient to have techniques focussed on specific biodiversity components (e.g., selected taxa), these are often inadequate surrogates for biodiversity (e.g., Andelman and Fagan, 2000; Possingham *et al.,* 2002). Selecting specific components also has limitations against a backdrop of unforeseen events that occur in the longer term. For example, in the 1970s there was concern about the impacts of red deer (*Cervus elaphus*) on regeneration of eastern South Island beech forests. A permanent-plot study was set up in Craigieburn Forest Park to investigate red deer impacts on mountain beech (*Nothofagus solandri* var. *cliffortioides*) regeneration. Fortunately, a wide variety of parameters were measured on the plots, including the frequency of exotic plant species. At the same time, nearby tussock grasslands were being invaded by exotic *Hieracium* species (e.g., Rose *et al.*, 1995). Many considered this invasion a consequence of land degradation brought about by repeated burning and sheep grazing. Yet, in the adjacent mountain beech forest, the permanent plots also showed a marked invasion of *Hieracium* over 23 years, calling into question the land degradation paradigm, since these forests have been neither burned nor grazed by sheep (Wiser *et al.*, 1998). The implication is that measurements focussed on contemporary perceptions and issues could easily constrain the future usefulness of monitoring information, and in the end influence commitment to forest biodiversity monitoring because of a lack of confidence in the data's applicability.

3. Build on the past

New Zealand's existing indigenous forest plot data provides a quantitative basis for detecting recent changes, as well as for designing new monitoring systems (e.g., Wiser *et al.*, 2001). However, because sampling intensity varied locally, analyses using many of these plots pooled into larger datasets can produce biased results for scale-variant relationships. There are strategies to overcome bias from uneven sampling intensities. Bellingham *et al.* (2000) used several thousand existing 0.04-ha plots to estimate unbiased biodiversity parameters for indigenous forests nationally, and to establish the number of plots required to quantify change in these parameters. An example parameter was the average basal area of the relatively common tree species kāmahi (*Weinmannia racemosa*) on existing plots across the area mapped as forest on the New Zealand Vegetation Cover Map (VCM) (Newsome, 1987). The method selected a random plot from near the intersection of evenly spaced grids, of varying density, superimposed on the Vegetation Cover Map. This gave a representative sample because existing plots were randomised in their location (see Allen, 1993). As grid density increased, proportionately fewer grid intersections had associated plots, although the total number of plots associated with grid intersections increased. The mean (and standard error) kāmahi basal area was calculated using tree diameter data from plots selected at the various grid densities (Fig. 1). At a national grid density of 8 km x 8 km (232 plots selected) the estimate of kāmahi mean plot basal area stabilised at about 7.1 m^2 ha^{-1} (Bellingham *et al.*, 2000). It was estimated that 1040 plots on a 7 km x 7 km grid nationally would be required to estimate kāmahi basal area to within 10% at the 95% probability level, although most of these plots would not contain kāmahi. Such analyses of existing data can be used to determine the sampling intensity needed to estimate various parameters, at given levels of precision, for various spatial scales [see Hall *et al.* (2001) for a carbon storage example]. Although the Land Cover Data Base (LCDB) (Dymond *et al.*, 2001) has recently superseded the Vegetation Cover Map, the principles outlined above still apply.

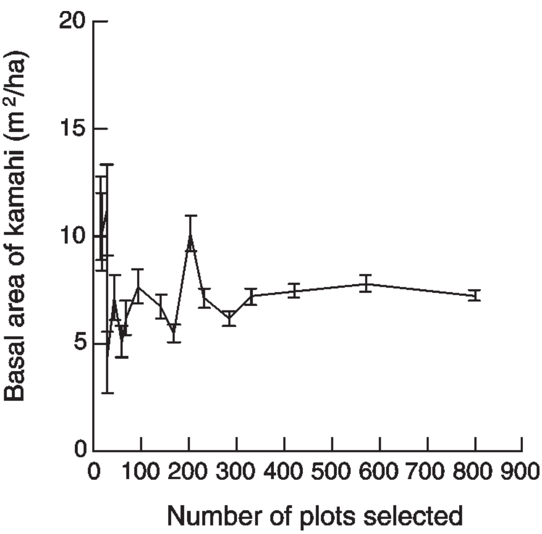

Figure 1. Basal area of kāmahi (*Weinmannia racemosa*) on 0.04-ha plots in forests, and associated standard error, for various numbers of plots selected nationally.

4. Make sure data are explicitly comparable

New Zealand has a long history of developing techniques for measuring structural and compositional changes in indigenous forests (e.g., McKelvey *et al.,* 1958; Wardle *et al.,* 1971; Meurk and Buxton, 1991; Coomes *et al.,* 2002). The merits and limitations of any approach are to some degree in the eye of the beholder, and work within an individual's view of what biodiversity components are important, what spatial and temporal scales are relevant, as well as what interpretations are possible. In the extreme, persuasive individuals have focussed resources on specific techniques through seductive claims of efficiency and simplicity, which may or may not have been so, and the use of some techniques died as the individual (or group) was promoted, changed interests, retired, or was overshadowed. However, it is important to ensure comparability of data over decades, and probably centuries. An overriding principle to achieve this would be to always build on the past. This does not necessarily mean we are left with the same techniques, but at least the data from a succession of techniques must be explicitly comparable.

5. Recognise the advantages of repeat measurements

When permanent plots are used, remeasurement will detect small changes in biodiversity parameters because repeat measures allow spatial variability to be partitioned from temporal changes. As Vanclay (1995) observed, "long-term studies are especially important for validation and to detect subtle trends, so 'old' plots and data should not be neglected but treasured". For example, data from historical plots have been used to infer effects of global warming on tree dynamics and carbon stocks in the tropics (Phillips and Gentry, 1994; Phillips *et al*., 1998). In New Zealand, existing plots cover most of the latitudinal range of forests, with many established 30 or more years ago, and many measured at least twice since establishment (Wiser *et al*., 2001). Coomes *et al.* (2002) showed that when such plots are re-measured over appropriate intervals, small changes in biomass carbon storage can be detected. They used power analysis to show that 600, 0.04-ha plots nationally would detect significant trends in living biomass of around ±0.03% per year. The same principles apply to biodiversity parameters. Although this reasoning strongly supports the use of permanent plots, where there are observer impacts on plots there may be advantages to sampling with partial replacement.

6. Account for difficulties in establishing baselines

The difficulties of defining historical baselines for assessing changes in biodiversity are increasingly recognised. There is little detail on these baselines before human-related impacts and also we know little about the changes that would have occurred in forests over the last millenium without human-related impacts. The dynamic nature of New Zealand's indigenous forests illustrate these difficulties. For example, a marked decline in northern rātā (*Metrosideros robusta*) and kāmahi in some North Island forests has been attributed to browsing by brushtail possums (*Trichosurus vulpecula*) (e.g., Rogers and Leathwick, 1997). It is also likely there has been considerable variability in the biomass of these tree species over recent centuries due to disturbance such as fires and volcanic eruptions (e.g., Payton *et al.*, 1984; Clarkson *et al.,* 1988); so a baseline selected from the past will not necessarily reflect the dynamics of these forests and where they would be today in terms of species dominance without the impacts of introduced herbivores. It is now unclear what the initial levels of dominance were in these forests. Paleo-ecological research shows that major changes of dominant trees in New Zealand forests often occur within a few tree generations (McGlone *et al.*, 1996) and there are few known instances of compositional stability in forests before human settlement. Even where conifer dominance was maintained in central North Island forests after disturbance by a volcanic eruption (Wilmshurst and McGlone, 1996), the forests buried were sometimes dominated by quite different species than those currently extant (Clarkson *et al.,* 1988). After relatively recent, complete loss of northern rātā and kāmahi from Ruahine Range forest canopies there has been a near absence of regeneration by these species (Rogers and Leathwick, 1997). Because of the long-lived nature of many tree species in New Zealand forests (e.g., >500 years for northern rātā) a return to pre-disturbance forest structure may be a millennium away — assuming such a recovery is possible in the face of seed dispersal limitation and modified understorey composition. Such examples support a view that there is no reason to expect forest management, such as culling browsing animals, will necessarily return forests to some state similar to that which would have occurred without human impacts (see Hobbs and Norton, 1996; Coomes *et al.*, 2003). In the face of pervasive human impacts, poor knowledge of historical baselines, and the likely irreversibility of some impacts, land management agencies must adopt more specific, forward-thinking management goals.

Table 1. Factors commonly shown to influence the 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 (modified from Allen *et al.,* 2003).

Factors	Variables	Examples of mechanism
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 effects	Litter quality	Modifies the abilities of seeds of other species to germinate and grow
Climate	Temperature, precipitation	Changes physiological processes
Soil	Texture, N availability, cation 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

7. Collect interpretive data

It has proven difficult to determine the cause of some measured changes in the structure, composition, and functioning of indigenous forest ecosystems. Such changes can be correlated with many factors (Table 1), some of which are relatively easy to determine (e.g., rainfall) and others we know little about (e.g., species effects on ecosystems). Often these factors are themselves correlated so that it is difficult to define causal relationships. A case study from South Island forests illustrates the value of collecting interpretive data. Extensive synchronous mortality of southern rātā (*Metrosideros umbellata*) and Hall's tōtara (*Podocarpus hallii*) in Westland conifer broadleaved-hardwood forests since the 1950s has been attributed to browsing by brushtail possums (Rose *et al.*, 1992). A study in south Westland evaluated tree mortality (supposedly reflecting herbivory) along a brushtail possum invasion front (Stewart, 1992). It was only once the study was completed that the invasion front was shown to also coincide with a soil fertility gradient related to plant species composition. Because, for some Westland forest species, the number of dead trees apparently covaries with soil fertility (see Reif and Allen, 1988), it was difficult to determine whether spatial patterns in tree mortality were simply the result of brushtail possum density varying along the invasion front or confounded with some other factor such as soil fertility. Soil fertility is often useful to measure because it is a strong determinant of biodiversity patterns in New Zealand indigenous forests (Wardle, 1991). As well as collecting data on possible confounding factors, there is considerable merit in collecting complementary interpretive data. For example, understanding impacts of introduced browsing animals on indigenous forest understoreys is clearer when diet data are also collected from the same locality (e.g., Forsyth *et al.,* 2002). This need for complementary data is also exemplified by forest plot designs where overstorey data are collected in a way that allows interpretation of understorey biodiversity changes (e.g., Allen, 1993).

8. Ensure there is long-term commitment

Franklin (1989) considered that long-term ecological studies (for which the motivation is in many ways similar to long-term monitoring) require a fortuitous combination of leadership, an opportunity and funding. The New Zealand experience supports his view; some of the best long-term biodiversity records result from the efforts of individuals, or small interest groups, who remain active over the long periods when monitoring is out of vogue. Although government departments have often funded forest biodiversity monitoring in New Zealand, it has been supported in a somewhat piecemeal fashion and major losses of long-term records have taken place during recent departmental restructuring. Given the importance of long-term studies in identifying and resolving environmental issues, Franklin (1989) asked whether we should tolerate such a 'serendipitous' approach. We believe that central and local government agencies must better support long-term biodiversity monitoring in a way that is co-ordinated to efficiently achieve their respective goals.

9. Ensure data storage and accessibility

The reasons for promoting appropriate data storage are to safeguard the long-term investment in environmental monitoring and to optimise the potential knowledge gains from these data (Wiser *et al.*, 2001). Arguably, research institutes, rather than government agencies, have shown the greatest commitment to storing large amounts of ecological data. One benefit for them of accumulating such data in a databank is to increase their ability to synthesise biodiversity distribution and dynamics across a wide range of scales (e.g., Wardle, 1984; Leathwick, 1995). The ability to combine data sets supports the use of standardised techniques, carefully described in manuals, and linked to data management systems that store the metadata and raw data in readily accessible, yet specific ways (e.g., see Allen, 1993; Wiser *et al.*, 2001). Standardised procedures will also minimise data variability caused by, for example, field teams having different levels of expertise. Much can be learned from National Forest Inventories throughout the world because they are well-established activities, designed and tested in operational use, and include quality assurance procedures for methodology, data handling and presentation (Larsson, 2001). Clearly the future lies in web-based data entry, with checking, query and analysis capacity that will meet both the needs of data providers and users. Effective implementation will require clear protocols for data access, use, acknowledgement, and cost recovery.

Designing a carbon monitoring system

Many of the principles addressed in this article arose during the design of New Zealand's Carbon Monitoring System (CMS) (Table 2). This system is currently being implemented by the Ministry for the Environment to report nationally and regionally on carbon storage in New Zealand's forests and shrublands, as required under the Kyoto Protocol. With input from a range of agencies (e.g., Department of Conservation, Ministry of Agriculture and Forestry), this system was explicitly designed to be multifunctional, so that field

Table 2. Assessment of the Carbon Monitoring System (CMS) against our set of principles.

Principle	Assessment
1. Define the problem and goals of monitoring	Goals defined at national scale by Kyoto Protcol, but regional scale carbon and other needs not well developed
2. Do not focus only on current perceptions	System measures fundamental characteristics and will provide a wide range of information
3. Build on the past	CMS built on the past by using existing data to determine sampling intensities
4. Make sure data is explicitly comparable	The CMS plot design extended a widely used method
5. Recognise the advantages of repeat measurements	Plots are permanently marked, with about one third being existing permanent plots
6. Account for difficulties in establishing baselines	Kyoto Protocol is explicit about 1990 baseline which was estimated using the best available data
7. Collect interpretive data	Appropriate set of interpretive data is being collected
8. Ensure there is long-term commitment	Although the CMS data is relevant to a wide range of agencies its implementation is largely funded by the Ministry for the Environment
9. Ensure data storage and accessibility	Not yet developed in any detail

measurements being made for carbon stocks also apply to biodiversity (Coomes *et al*., 2002). The Ministry for the Environment is largely funding implementation of the CMS although other agencies will strongly benefit. The system is based on periodic updates of the LCDB, as well as systematically located permanent plots on an 8 km x 8 km grid throughout the country's indigenous forests and shrublands. The LCDB is evolving, with a current emphasis on detecting the extent and spatial location of land-use changes as well as greater discrimination of types within forests and shrublands (Dymond *et al.*, 2001). To provide continuity, and build on the past, the CMS has been designed so that about one-third of plots are existing ones matched to nearby grid intersections, and the rest are new plots established at unmatched grid intersections (Fig. 2). The plot measurements build upon the 20 m x 20 m quadrat method (area 0.04 ha) that is widely used in New Zealand (Allen, 1993). A consequence of this design is that it optimally uses past plot data but extends coverage to previously unsampled or undersampled localities. It is important to sample the geographic distribution of forests because of, for example, regional differences in disturbance regimes. To provide an independent review, and international acceptability, the CMS was reviewed and endorsed by an international review panel (Theron *et al.*, 1999).

Biodiversity monitoring in the future

Future biodiversity monitoring will be likely to include a combination of remote sensing and point-based sampling and modelling (e.g., Nagendra and Gadgil, 1999). Remote sensing offers complete spatial coverage and efficiency but, as yet, is incapable of capturing the full range of compositional, structural and compositional information required. Remote sensing will contribute spatial coverage information, with increasing resolution over time, and point-based sampling (commonly as surveyed plots) will give the detail necessary on a wider range of characteristics (Larsson, 2001). If plots are used to ground truth remote sensing methods it is important that both measure biodiversity at the same spatial scales and also that spatial locations are precisely matched. Modelling can generalise, with varying levels of precision, some monitoring results in time and space.

Sampling design

A challenge in designing strategies for sampling biodiversity is to balance national needs with those pertinent at a local scale. Overton and Stehman (1996) pointed out that the framework for long-term, spatially extensive monitoring must be updated periodically, and new issues addressed. They recommended that any system should not be based entirely on current perceptions and needs. For example, the perceptions of whether forests are more appropriately stratified based upon current administration boundaries, types of forest, environments, catchments, phylogenetic distinctiveness, or disturbance will change over time according to purpose. In addition, Overton and Stehman (1996) considered that there are statistical advantages from a simple design that will, over time, compensate for possible precision or efficiency lost by not using more complex designs (e.g., pre-stratification). As a consequence, we do not support any form of

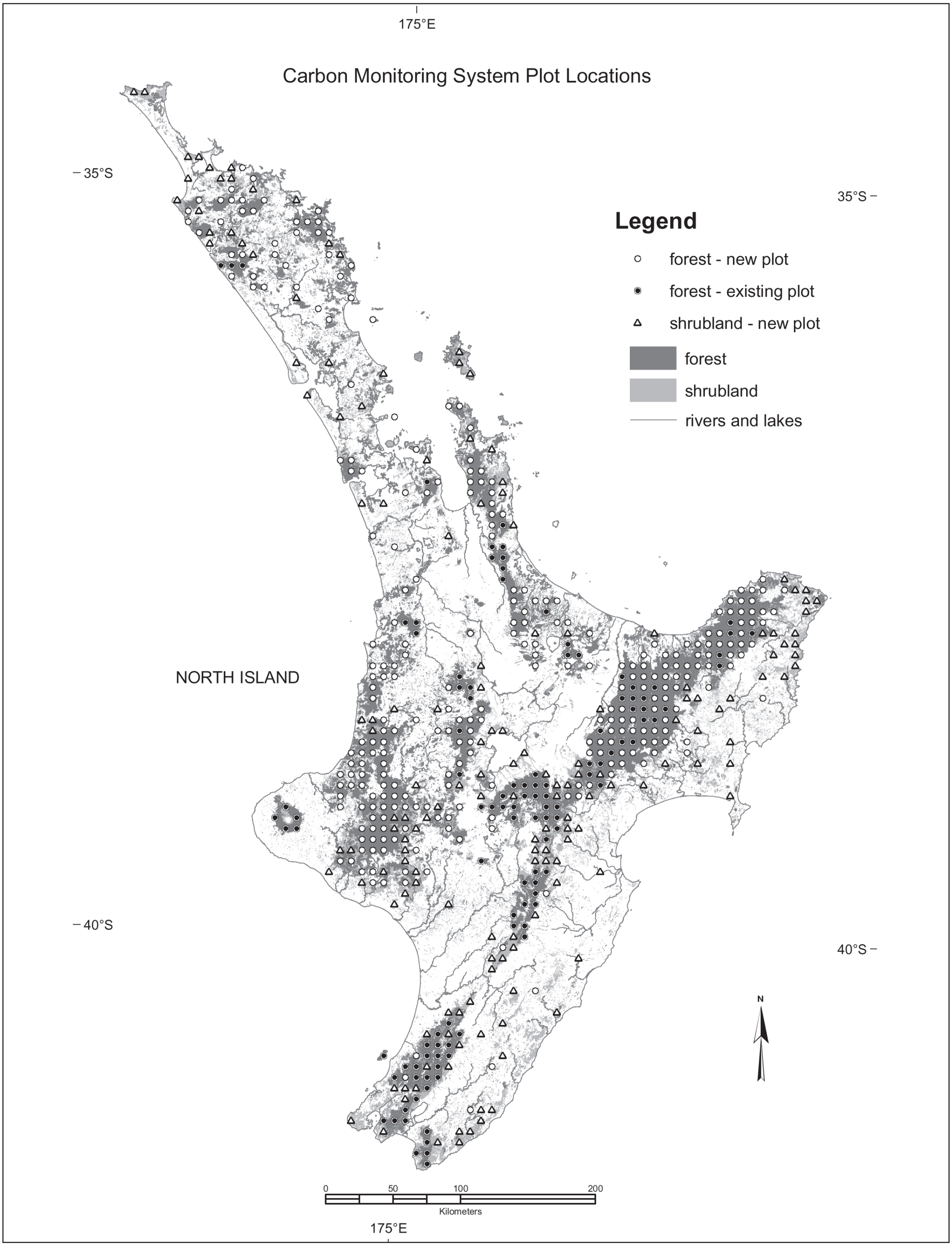

Figure 2. Presence or absence of existing permanent 0.04-ha plots nearby to points on an 8 km x 8 km grid superimposed on New Zealand forest and shrubland area from the Land Cover Data Base. This design is the basis for New Zealand's forest and shrubland Carbon Monitoring System. South Island Plots are shown on the following page.

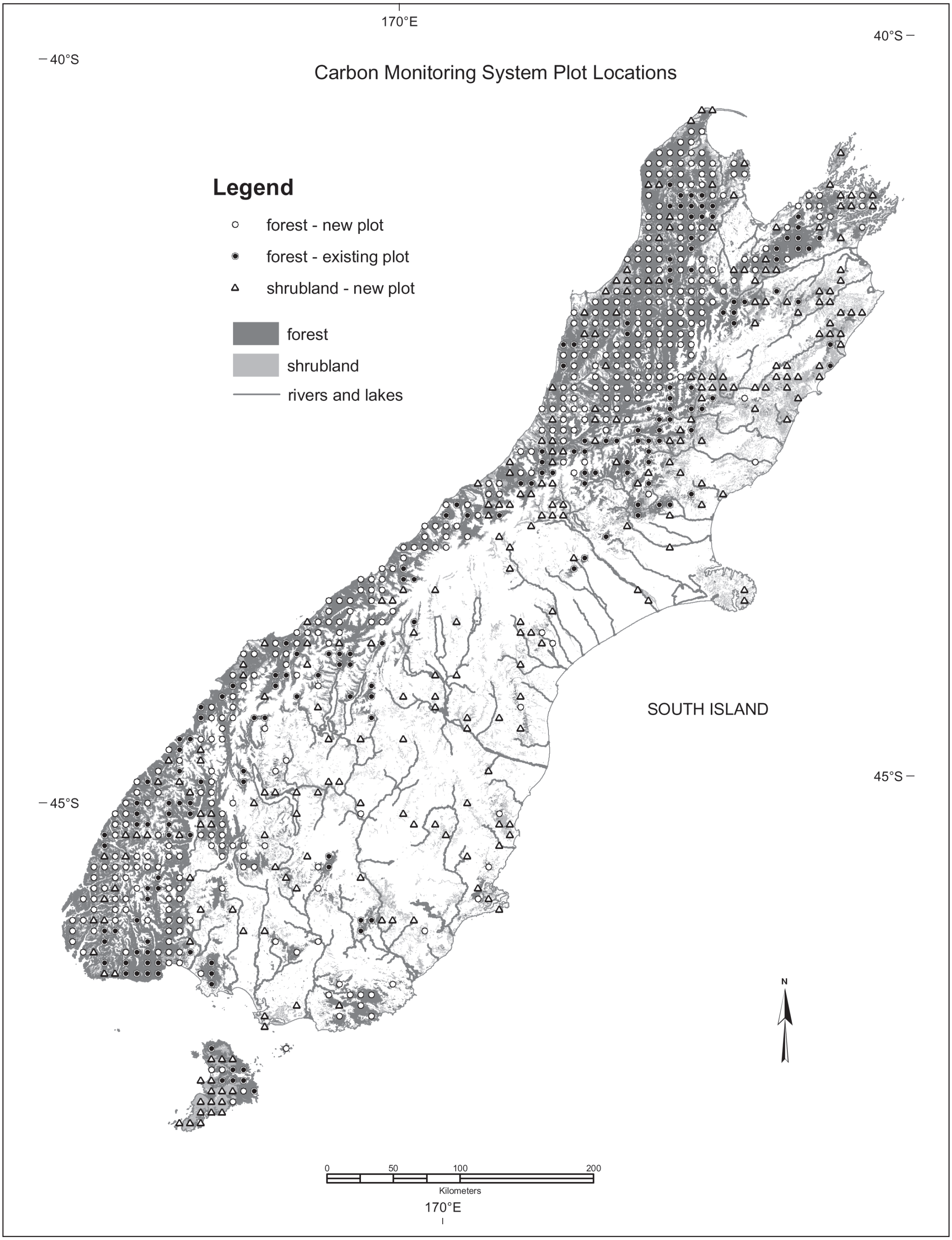

Figure 2 (continued). South Island CMS plot locations.

stratification for long-term monitoring. In reality, it is impossible to comprehensively test this view because, for example, it is too difficult to anticipate all future uses of biodiversity monitoring data. However, it is for such reasons many countries, such as the U.S.A. (Stolte, 1997), Austria (Monserud and Sterba, 1996), Finland (Tomppo, 1998), and Canada (Barker *et al.*, 1996), employ grid-based, systematic designs for plot locations in national-scale forest monitoring systems. Systematic designs often give better spatial coverage than random sampling. For long-term, spatially extensive, national and regional coverage, there is considerable merit in an unbiased, unstratified sample of indigenous forests.

We have already discussed how individual plots, within local plot networks, can contribute to larger-scale monitoring. Local networks, with higher sampling intensities, are one way to provide the second tier to a national network, enabling finer-scale resolution of parameters (Bellingham *et al.*, 2000). We advocate retention of many of these local plot networks, spanning as great a geographic range as possible. Particular value should be placed on those plots maintained over long time periods. Where such networks have subjectively located plots, or sampled strata (e.g., based on canopy disturbance), their use in some larger-scale monitoring can be limited (Bellingham *et al.*, 2000). We also advocate establishment of new local plot networks, possibly using a higher grid density than that used for national coverage, in areas where existing coverage is poor, such as in Northland or inland Taranaki (Wiser *et al.*, 2001). In some instances local networks are foci for research elucidating the mechanisms behind trends observed nationally or regionally, and this focused effort parallels long-term ecological research sites that have been established in many countries (Likens, 1989).

There will always be rare habitats and environments where the sampling intensity for general monitoring is inadequate to quantify change within desirable levels of precision. Certainly grid density can be altered. In the Canadian forest monitoring system, grid densities vary from 4 km x 4 km to 80 km x 80 km, where the low-intensity sampling points are a subset of the more intensive sampling points. Varying sampling intensity would also then influence the number of historical plots used because increasing intensity would result in more historical plots being selected. As monitoring systems are developed for greater spatial resolution it is easy to develop systems that become too onerous for a given level of precision. The grid, with varying sampling intensities, that we have described should be extended to cover the entire New Zealand landscape with adequate sampling of forests but without giving them special treatment.

What to measure

It is important to understand why characteristics measured in a certain way are useful and repeatable, and what needs to be measured to help interpret any changes. It is most efficient if these measurements satisfy a range of purposes. As an example, the U.S. forest health monitoring system addresses national concerns (e.g., pollution effects) but at the same time it meets international requirements such as those under the Montreal Process (Stolte, 1997). The U.S. system involves a co-operative, multi-agency programme to monitor, assess, and report on the status, changes, and trends in forest ecosystem health and sustainability. In Europe, a project on monitoring and evaluation of forest biodiversity brings together 19 partner countries representing six major biogeographic regions (Larsson, 2001). Bellingham *et al.* (2000) support a consortium approach in New Zealand involving a range of government agencies and drawing upon experience elsewhere in the world.

The U.S. forest health monitoring system involves satellite monitoring to provide evaluations of forest fragmentation and use; aerial monitoring to detect local problems, such as pest outbreaks; and ground-based plots to provide data on forest structure, diversity, and other site-specific indicators (Table 3). Specific national concerns in the U.S.A. include the ongoing impacts of atmospheric pollution, which is not of wide-scale concern in New Zealand. Forest managers in New Zealand have their own concerns currently focussed on the impacts of introduced pests and weeds. Increasingly management of forests in New Zealand is being influenced by international requirements (e.g., Kyoto Protocol, Forest Stewardship Council Certification, Montreal Process). In the following discussion we suggest some broadbased indicators that would partly satisfy New Zealand's current forest biodiversity monitoring needs for regional, national and international requirements (for more detail see Allen *et al.,* 2003):

Table 3. Forest health monitoring indicators currently in use, or that have been tested, within the U.S.A. at national and regional levels.

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	

1. Forest area

Woody plant communities are expanding as extensive agriculture in low productivity environments becomes unprofitable. Because New Zealand had a forested pre-settlement landscape, much of the landscape could eventually develop towards forest. McGlone (2000) argued that, if correctly managed, this expansion is a major opportunity for the conservation of biodiversity in New Zealand at a landscape scale. It is important to quantify the area changes, the spatial distribution, as well as the structure and composition of these "new" indigenous forests as they develop. Biodiversity within these "new" forests would be captured in a monitoring design that allows for sampling of the entire New Zealand landscape. The LCDB is based upon satellite imagery and gives a 1-ha spatial resolution of land cover types that could allow indices of forest fragmentation, such as size, isolation, shape, and edge:area ratio (e.g., see Norton, 2000), to be determined; this can be done at a range of spatial scales above one hectare. Field data will also be required to quantify the compositional, structural, and functional characteristics of forests as outlined in the following sections.

2. Tree mortality and recruitment

Because trees drive many of the compositional and functional changes in forests, it is important to measure and understand patterns of tree mortality and recruitment. Globally many tree species are threatened by human impacts (Williams, 1998). The maintenance of tree species can be viewed as a population process and can be described and modelled on the basis of comparing mortality and recruitment rates (Sheil *et al.*, 1995).

Bellingham *et al.* (1999) examined the imbalance of mortality and recruitment of dominant tree species at 14 localities around New Zealand using tagged individuals on permanent plots. Mortality rates of Hall's tōtara were markedly higher than recruitment rates in four of six localities where this species occurred. This indicates we should be concerned about the viability of Hall's tōtara populations. In a general sense, the species of most concern should be those with high mortality and low recruitment at many localities, and these should be the focus of research to explain the paucity of regeneration and of restorative actions (Allen *et al.*, 2002). For such purposes, permanent plots should be of sufficient size to characterise tree mortality and recruitment, yet it is surprising some authors suggest plots as small as 25 m^2 to monitor tree population dynamics (e.g., Sweetapple and Burns, 2002).

3. Exotic plants

We need to understand how widely distributed exotic species will become and how they will impact on forests over time. For *Hieracium lepidulum* invading a 10 000 ha Canterbury mountain beech forest, Wiser *et al.*, (1998) showed that our ability to predict its distribution increased during the invasion, but also that the site variables that best predict a weed's distribution changed with time. The network of permanent plots established by the U.S. forest health assessment uses the percentage of species that are exotic as an indicator of exotic invasion (Stapanian *et al.*, 1998). Such an indicator requires compositional data on indigenous and exotic species rather than a focus on specific taxa. Compositional data from permanent plots is highly desirable. For example, the impact of exotic invasions can be assessed through associations with changes in tree or understorey herbaceous species.

4. Browsing impacts of introduced animals

Introduced animals have pervasively modified indigenous forest understoreys with a wide range of implications for both above-and below-ground processes (e.g., Wardle *et al.*, 2001). Assessments of leaf browse have commonly been used to assess impacts, and have been extensively tested and used over the last 30 years (Wardle *et al.*, 1971; Rose and Burrows, 1985; Payton *et al.*, 1999). For example, 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. These browse indices are derived from easily-made observations of plant species browsed on plots. Procedures must be carefully standardised among observers over time. Because the observations are made on foliage, any browse observed is a short-term impact, relative to the periodicity of plot remeasurement. With appropriate statistical methods, such indices could be estimated on tagged individuals (e.g., trees), and make a long-term contribution to linking defoliation and demographic parameters.

5. Quantity of dead wood

Dead wood performs many important ecological functions: providing habitat for organisms; playing a part in energy flow, carbon and nutrient storage; and influencing sediment transport and storage (Harmon *et al.*, 1986). 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 and O'Donnell, 1999). Saprophytic fungal communities are diverse in logs on the forest floor, even in apparently simple mountain beech forest, and play a role in nutrient transport from logs to the soil (Allen *et al.*, 2000). Many studies have shown the importance of logs on the ground for the establishment, growth, and survival of tree seedlings. 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 individuals have been measured, and the logs categorised in decay classes (e.g., Coomes *et al.*, 2002).

Conclusions

The impetus to assess the status of New Zealand's indigenous forests has possibly not been so pronounced since the National Forest Survey of the 1940s and 50s. This is reflected in initiatives such as New Zealand's CMS, which can now also provide the first unbiased national estimates of some biodiversity parameters (e.g., exotic weeds). The scientific community needs to agree on the technical aspects of such initiatives and support their implementation, and all relevant policy and management agencies must unite to support their ongoing maintenance. At present, the Department of Conservation, Ministry for the Environment, Ministry of Agriculture and Forestry, Statistics New Zealand and local government agencies have overlaping requirements from biodiversity monitoring and these would be best met through co-ordinated effort. Conservation managers will benefit because they will have a more robust basis for determining changes in indigenous forest biodiversity in response to management. The resulting databases will also allow researchers to increase our understanding of forest dynamics as a basis for improved management. These considerations are most developed for forests but the design is readily extendable to a wide range of other ecosystems. Certainly there is scope for monitoring a wider range of organisms than considered in this paper (e.g., see Larsson, 2001). Although the type of design we have discussed will be suitable for some of these, such as litter invertebrates, it will not be suitable for others, such as birds.

Without spatially extensive, robustly designed, biodiversity monitoring systems, the country will remain in a relatively weak position nationally, and internationally, to report on the effectiveness of biodiversity conservation. This could impact adversely on New Zealand's standing in the global community and ultimately influence our access to international markets and trade.

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