

Potential value of indicator species for conservation and management of New Zealand terrestrial communities

SCIENCE FOR CONSERVATION: 109

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Published by
Department of Conservation
P.O. Box 10-420
Wellington, New Zealand

Keywords: Indicator species, biodiversity, conservation, vegetation, vertebrates, invertebrates, hierarchical system research, standardisation, system management

Science for Conservation presents the results of investigations by DOC staff, and by contracted science providers outside the Department of Conservation. Publications in this series are internally and externally peer reviewed.

© April 1999, Department of Conservation

ISSN 1173-2946

ISBN 0-478-21817-6

This publication originated from work done under Department of Conservation Investigation no. 2242, carried out by John Hutcheson and Patrick Walsh, Forest Research Associates, PO Box 1031, Rotorua and David Given, David Given & Associates, 101 Jeffreys Road, Christchurch. It was approved for publication by the Director, Science & Research Unit, Science Technology and Information Services, Department of Conservation, Wellington.

Cataloguing in Publication

Hutcheson, John (John Alan)

Potential value of indicator species for conservation and management of New Zealand terrestrial communities / J. Hutcheson, P. Walsh, D. Given. Wellington, N.Z. : Dept. of Conservation, 1999.

1 v. ; 30 cm. (Science for conservation, 1173-2946 ; 109.)

Includes bibliographical references.

ISBN 0478218176

1. Biological diversity conservation--New Zealand. 2. Ecological surveys--New Zealand. 3. Vegetation surveys--New Zealand. 4. Invertebrates--New Zealand. I. Walsh, P. (Patrick Joseph), 1953-. II. Given, David R. (David Roger) III. Title. IV. Series: Science for conservation (Wellington, N.Z.) ; 109.

574.993 20

zbn99-030752

Contents

Abstract	5
<hr/>	
1. Introduction	6
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1.1 Report brief	6
1.2 Objective	6
1.3 Report structure	6
2. Overview	7
<hr/>	
2.1 Introduction	7
2.1.1 Nature and extent of biodiversity	9
2.1.2 Interactions and the dynamic nature of biotic systems	9
2.1.3 Problems of perceptions and scale	11
2.1.4 Lack of baseline knowledge	13
2.1.5 The need for a systems-based approach	14
2.2 Current approaches to selection and use of indicator species	14
2.2.1 The limits of change approach	14
2.2.2 Characteristics of indicator species	15
2.2.3 Baseline studies and monitoring	18
2.2.4 The need for integrated monitoring	18
3. Vegetation	19
<hr/>	
3.1 Introduction	19
3.2 An hierarchical approach	19
3.3 Rapid standardised documentation—Level 2 studies	21
3.4 Indicator value of species richness	23
3.5 Level 3 studies	24
3.5.1 Demographic monitoring	24
3.5.2 Resampling frequency	26
3.5.3 Non-flowering plants	30
3.6 Genetic concerns	31
3.7 Recommendations for progress in New Zealand	31
4. Vertebrates	33
<hr/>	
4.1 The unique nature of the New Zealand situation	33
4.2 Experience outside New Zealand	35
4.3 Conclusions	37
5. Invertebrates	38
<hr/>	
5.1 Introduction	38
5.2 Use of invertebrates for indicating and comparing biodiversity	39
5.2.1 Insects: the heart of biological diversity	40
5.3 Potential indicator taxa	41
5.4 Sampling	43
5.4.1 Sample collection	43
5.4.2 Timing of sampling	48
5.4.3 Level of taxonomy required	48

5.5	Malaise trapped beetle samples as biodiversity indicators	49
5.6	Analyses of insect community data	50
5.6.1	Diversity measures	51
5.6.2	Species abundance patterns	53
5.6.3	Multivariate techniques	54
5.6.4	Use of classification to refine sampling protocols	56
5.6.5	Functional (trophic) structure	57
5.7	Conclusions and recommendations for New Zealand	57
6.	Conclusions	59
6.1	Indicators in the New Zealand situation	59
6.2	The recommended approach: an hierarchical model	59
6.3	The need for standardisation	62
7.	Acknowledgements	63
8.	References	63
<hr/>		
Appendices		77
<hr/>		
Appendix 1. Selected examples of vegetation monitoring		77
Appendix 2. Example of the 'Vulnerability Index'		90

Abstract

The use of indicator species is reviewed within a context of the need for broader understanding of biological systems and the effects of management on biodiversity. Current theory and practice is used to evaluate approaches to the knowledge gaps that exist. It is concluded that the understanding needed to guide appropriate land management requires the integrated study of communities and systems in order to derive the basic principles operating. Standardisation would enable efficient use of resources through an increased ability to collate and compare different studies.

Information gathered needs to be related to vegetation systems, as this is the level at which perception and management of the environment occurs. However, the great majority of terrestrial biodiversity occurs as insects, and so these must form a major part of any meaningful evaluation of biodiversity. Studies of vertebrates benefit from a context of system knowledge, but do not themselves provide this. The range of management questions, and the immense range and dynamic nature of biodiversity mean that standardisation must be of method rather than of specific organisms.

An hierarchical approach to problem evaluation is suggested as being appropriate, using methods which are currently available. Level 1 involves mapping of vegetation systems within landscapes. Level 2 provides descriptions of selected assemblages within recognisable systems, using standardised (and therefore comparable) and practical methodologies. These include the use of recce plots for characterising vegetation types, and Malaise trapped beetles for characterising invertebrate communities. Many broad management concerns may be able to be elucidated at this level. Level 3 represents the more intensive studies targeted at individual species using a broader range of approaches.

This hierarchical approach has already proven to deliver effective guidance for conservation management in New Zealand, and is in accord with current strategic planning. It enables a defining of questions and formulation of hypotheses from a broader context, allows results of studies to be interpreted within a landscape and system context, and to be collated and compared at various levels of resolution. The approach will enable mechanistic models to be proposed from data subsets, and then tested for validity across a range of systems.

1. Introduction

1.1 REPORT BRIEF

The brief of this report was to assist the Department of Conservation (DOC) in creating an information base on which to design valid research into selecting species or taxa which could be used to monitor terrestrial habitat quality in areas subject to management.

1.2 OBJECTIVE

The objective of this review is to place relevant research on indicator species, and the methodologies that could be used to monitor terrestrial habitat quality and biodiversity, in a New Zealand context. Selected information has been compiled on the uses, value and appropriate application of indicator species in both managed and natural terrestrial ecosystems to help DOC in the design of relevant research to maximise habitat management and protection.

Guidelines indicated that the review was to assist DOC in making some judgement on the scientific validity of using indicator species for studying terrestrial communities. The emphasis was to be on invertebrates and appropriate methodology in natural systems. Approaches were required which have the potential for assessing habitat management success, and for identifying areas of high biodiversity and endemism (G. Sherley, DOC S&R, pers. comm.).

1.3 REPORT STRUCTURE

The report begins with an overview of biotic systems, their inherent dynamic nature, and our attempts to better understand them through the use of indicators. The need for understanding at scales of management and systems are discussed, and a summary of traditional approaches to the selection and uses for indicator species is included. Following sections of the report provide examples and discussion on plants, vertebrates and invertebrates, respectively. The authors found the subject matter difficult to separate into discrete components, reflecting the interrelationships inherent both within biotic systems, and with the concept of using indicators for broader interpretation of systems. Some information is included from managed production systems, as these have received greater research attention. Overseas research is interpreted within the New Zealand context. The conclusion identifies promising approaches for research in New Zealand. Lessons from a series of successful vegetation studies are appended.

2. Overview

2.1 INTRODUCTION

The value of the world's biological resources has been estimated at about 33 trillion dollars, or twice the world's total annual GNP (Costanza et al. 1997). However the true value is infinite, as human society cannot exist without them. All these resources are now 'managed' in the sense that all ecosystems on earth are affected by human activity. Therefore perhaps the most important questions currently facing our society are 'what happens to these biological resources as we manage them?' and 'how can we sustain both human society and the irreplaceable resources of biodiversity?' The difficulties in answering these questions arise from three areas: the extent of the resources; their dynamic nature along nearly limitless gradients of space, time and taxonomic scales; and our general lack of baseline knowledge.

Biodiversity refers to genetic variation, and sustaining a resource refers to its retention. There may be great complexity involved in sustainable management of biodiversity, but our children can understand the basic concepts. There are many situations in which sustainability is obviously not occurring. In many land-based production systems, management inputs have replaced ecosystem functions (e.g. nutrient cycling being replaced by fertiliser inputs). In many conservation efforts the functions of the wider biotic community have been replaced by e.g. captive breeding. Neither situation is regarded as sustainable. However, improved knowledge of systems can assist in understanding how to better achieve management aims with reduced management input.

All species have particular environmental requirements. Their relative success, measured as the abundance of a given population, is indicative of a combination of all these conditions. Some species have particular attributes (e.g. narrow tolerances of some parameter) that may enable them to signal changes in ecosystems. Such species have often been selected for monitoring environmental conditions. Which species are selected depends on the questions being posed and especially whether one is monitoring one aspect of an ecosystem for a specific purpose (e.g. water quality under the Resource Management Act (RMA), or one or more endangered species) or broader qualities such as health or biodiversity of systems. In many instances, indicator species have been found to reflect the quality and changes in environmental conditions as well as aspects of community composition.

Indicator species are generally selected from whatever taxonomic group fulfils the needs of the immediate management problem. This has led to a plethora of organisms which have been found to be useful, e.g. amphibians, molluscs, birds, corals, beetles, aquatic flies, fungi, and marine invertebrates. Population characteristics monitored include distribution, abundance, and demographics; reasons for selecting particular indicator species may commonly include ease of sampling and identification and their use as a resource, quite apart from their

indicator value. Species that are chosen should provide standards of consistency and precision about changes in a community and they should be reliable enough that changes and trends are unambiguously detected (Global Biodiversity Assessment (GBA) 1995).

Indicator species (or groups of indicator species) are involved with information extrapolation, i.e. in revealing something extrinsic to the data collected. Indicators should tell us something about the environment which is not readily apparent, or which cost prevents us from measuring directly.

While each individual use of indicator taxa may be justifiable within the parameters of the immediate management concern, there are two points that should be considered:

- (1) The interrelationships within communities means that species responses are driven by influences that are multi-factorial in origin. This may lead to confusing interpretations when based on information from too narrow a portion of the system.
- (2) There is a need to conserve resources through the ability to collate and compare studies. This requires a standardisation of a core set of methodologies across studies of different phenomena.

A good example of the interrelationships within biotic systems, and of species groups indicating changes in the environment which were not readily apparent, can be found in research being carried out at the Slovakian Academy of Sciences (J. Novotný, Slovak Republic Forest Research Institute pers. comm. 1996).

Research indicated that vertebrate numbers were falling in a forest close to the city of Zvolen. Investigations of this phenomenon showed that although insect herbivore numbers were ten times higher than in other forests of the region, the vertebrates, and especially birds, were almost gone. Analysis of nutrients in the trees demonstrated that although there were elevated sugar levels in the foliage there was a lack of calcium. Although the insects were growing well and reproducing at a high rate they were not sequestering calcium. Birds which fed upon these herbivores failed to reproduce successfully. Most nests were unsuccessful and newly emerged chicks died due to this nutrient deficiency.

Further investigation revealed that the cause of this phenomenon was a nearby aluminium factory which had been operating with east European technology. The high aluminium content emissions were locking up calcium in the soil, resulting in this bottom-up degradation of the environment. Cleaner technology, from France, has now been installed in the factory. The herbivore load is being compared with unpolluted forests to measure improvements in the forest ecosystem. During 1996 the first breeding birds for several years were found within the forest and although the chicks did not survive, it was seen as an indication that environmental factors were improving. These studies are continuing to utilise the insect herbivores to indirectly measure improvements in plant quality and to use the birds to indicate the sequestration of calcium by the herbivores. This example shows that while vertebrate response was most noticeable, explanation of the causal mechanisms came from investigations at lower trophic levels (i.e. at the level of the plants and insects).

2.1.1 Nature and extent of biodiversity

Although most biomass is provided by vegetation, most of the terrestrial genetic variety at the species level is provided by insects (Southwood 1978a, Erwin 1983, Stork 1988, 1993, Wilson 1988, Gaston 1991, Hodkinson and Casson 1991, Hammond 1992, Kim 1993, Gaston and Hudson 1994, Watson et al. 1995). Species richness appears to peak close to an organism size of about 1 mm. This is apparently because organisms of smaller size tend toward higher dispersability of propagules, so there is no reason to believe that a plethora of microscopic organisms exist which will eventually outnumber the species of insects (Fenchel 1993). Genetic variation within species is also greatest for the insects, surpassing even that of plants (Berry 1982).

Species separations represent 'agreed' divisions along genetic gradients (O'Hara 1994), and there are cogent arguments for documentation of some selected biota (e.g. crop plants) at finer genetic scales than species. However, this is obviously inappropriate for descriptions and comparisons of natural ecosystem biodiversity, where we do not yet even know the majority of species relationships. The magnitude of the task of understanding biodiversity requires the use of tools that are most readily available. Published knowledge has traditionally been linked to species, and interpretation of the wealth of information that biodiversity represents is dependent on this published background. Much of the information sought relates to the unique qualities of the entities (e.g. endemism), and this knowledge may be available only at the species level.

The basis of the tools required for the task at hand, in the form of natural history collections and training of taxonomists, are currently in a lamentable state of neglect world-wide due to a lack of funding. This is despite the fact that these tools provide the means to interpret information on biodiversity, and to guide recognition of components of particular use (Cotterill 1995). Robert May, former physicist turned ecologist and now chief scientific advisor to the UK government, has drawn attention to the 'lunatic' inequities of funding between systematic taxonomy and astronomy. He notes that survival of human society is dependent on maintaining the integrity of the biosphere, and we appear to be on the breaking tip of the sixth big wave of extinction of life forms. In contrast, the stars will be largely unchanged and available for study 1000 years from now (Anderson 1997).

2.1.2 Interactions and the dynamic nature of biotic systems

Systems accumulate organic material following a successional trend toward greater organic structure. Time scales, trajectories and structures at any point in time are all influenced by combinations of abiotic and biotic factors in the local environment. These factors all change continually, on almost endless gradients of time, space and taxonomic constituents. Thus single-factor analyses are insufficient for understanding how communities are structured (Dunson and Travis 1991; Hunter and Price 1992, Polis 1991, 1994, Power 1992).

Mutualism

No species exists in isolation, and species only persist while individuals can withstand the local abiotic extremes and can suitably utilise the surrounding environment, including interactions with the local biotic community.

Competition for resources between component individuals in populations is overlaid on a ubiquitous background of mutualistic interactions between various functional groups that form, and benefit from, the system environment. This background of mutualism between different functional units is exemplified by the integration of mitochondria into other single-celled organisms, which fostered the evolutionary expansion of multicelled life (Margulis and Bermudes 1985). Innumerable mutually beneficial interactions exist in communities, e.g. the relationship of mycorrhizal fungi with most vegetation (Allen 1991), nutrient cycling by invertebrates, seed spread by birds, and the symbiotic relationships which result in coral reefs, lichens and ruminants. They are also apparent in the vast array of organisms which share the environment humans create. These include our hundreds of gut bacteria, our farm animals, pets, crops and weeds, urban bird life and the scavengers that live on our refuse.

System engineers

Individual organisms modify or 'engineer' their environment in various ways (Jones et al. 1994) contributing to the system environment, and thereby influencing its suitability for particular organisms. This is easily recognised at some scales. For example, temperature extremes are attenuated under New Zealand indigenous forest canopy. This may result in e.g. mean temperatures over a summer period being 2.3°C higher within the forest than in the adjacent farmland (Hutcheson 1996). The forest environment also retains the higher and more stable humidity necessary for decay processes and understorey plants, especially ferns. In a Northland forest, a combination of goat damage to forest edges and possum damage to the forest canopy allowed wind penetration resulting in loss of previous fern diversity (Hosking et al. 1989). Wind is a major influence on New Zealand vegetation. An example of this is provided by a study of isolated mountain beech stands in the northern Ruahines. This found stands to be 'migrating downwind', through breaking down on the exposed windward side, and regenerating on the sheltered side (Hosking et al. 1993). Prognosis for survival of the stands depended upon the local topography, i.e. whether stands were moving into a more exposed, or more sheltered site.

Modification of the local abiotic environment may occur at all scales and modes of existence, including that of the detritivorous insects, fungi and microbes involved in the recycling of organic material. These processes lead to an alteration of the nature of the soil (e.g. Willis et al. 1997). Workers attempting to model even 'minimalist caricatures' of system engineering by selected species find little real appropriate data to use in their calculations (e.g. Gurney and Lawton 1996). But all species may be thought of as 'system engineers' in that they all contribute to the processes occurring within systems. The lack of recognition of the wide variety of system engineers appears largely derived from a narrow perception of what constitutes a system engineer.

The extent of interactions in systems is far greater than that reported by much of the food web theory literature (Polis 1994). Therefore while single-species information may be impossible to extrapolate to a system context (Hellawell 1977) documentation of assemblages within systems can enhance our ability to interpret system status and processes, and to provide indications of the relative influence of management effects against this dynamic background (e.g. Hutcheson 1996).

Recent information suggests that dynamic processes within systems exert a strong influence on diversity of both plants (Attiwill 1994, Reice 1994) and insects (Hutcheson 1990, 1996, unpublished data, Hutcheson and Jones in press, Hutcheson and Kimberley in press). Such findings highlight the potential to confuse interpretation of sampled species richness from different stages in system dynamics of systems (e.g. stable forest cf. unhealthy or windblown forest) where this dynamic nature of the habitat is not recognised and documented.

Keystone species

Reference to keystone species have become commonplace in the literature as researchers identify components that are found to be crucial to the functioning of the various systems under study (e.g. Menge et al. 1994, Tanner et al. 1994). The term was coined for species whose effects in ecosystems are disproportionately large for their abundance or biomass (e.g. Paine 1966, 1974). It has also been used for species that form major components of systems, and for functional groups providing resources, e.g. epiphytes in rainforest (Nadkarni 1994).

The keystone theory has been suggested as a tool for resource managers (see Elnor and Vadas 1990, Foster 1990), on the assumption that keystone species controlled community structure over wide regions of time and space. Attempts to better define keystone species, and to objectively determine them, has led to a revision of the earlier acceptance of their generality, importance or even existence (Underwood and Denley 1984, Estes and Harrold 1988, Elnor and Vadas 1990, Foster 1990, 1991). Beck (1997) in discussing the keystone concept agrees with Elnor and Vadas (1990) that the assumed generality of the theory stifled critical research into whether this was warranted. The patch dynamic nature of terrestrial systems leads to the influence of different species and assemblages varying over time and space. Where this system context is not taken into account, localised studies may return conflicting conclusions as to the importance of various components.

2.1.3 Problems of perceptions and scale

How a community is measured not only defines our perception of the community, but also influences our ability to measure community response to extrinsic or intrinsic disturbance.

Broad management aims, such as sustaining qualities of biodiversity, immediately direct us to the need for management of entire systems. However, an obstacle to sustainable management is the narrow range of our perception of these entities and their dynamics. Observational scales determine the patterns that may be detected, and all scales have difficulties, both in interpreting information that is under the influence of factors at larger scales, and with the increased resources required at scales of increasing taxonomic resolution. The scale of examination chosen is thus fundamental to what can be observed, and also to the usefulness of the knowledge gained (Lawton 1996).

We perceive (and manage) systems in terms of the plant associations (i.e. vegetation systems) and these provide indications of the physical parameters of the environment. These systems integrate combinations of temperature, water

availability and edaphic factors, allowing us to recognise and document bioclimatic zones. They also reflect biotic factors, including effects of the vegetation itself, and anthropogenic influences, such as domestic stock. Vegetation systems provide the basis of our present indicators of the status of biological communities, and it makes good pragmatic sense to continue to do this. Vegetation system distribution can now be relatively simply documented and mapped to large scales in a GIS environment, using remote sensing in combination with ground surveys. Putative relationships within and between these systems may be extrapolated in a testable way at larger spatial scales by stratifying vegetation systems in landscapes.

However, vegetation systems in themselves are insufficient to answer questions about qualities of biodiversity. Rare endemic vertebrates have become the most common 'flagship' species for conservation effort in New Zealand, and these are also influenced by their system context, e.g. through food resources and predation by introduced vertebrates. In addition, the insects provide the overwhelming majority of terrestrial biodiversity, and therefore little can be said about the biodiversity of any terrestrial system without information on the insect component.

While recognising that our normal scales of perception form a narrow window on system processes, it is important to realise that some scales of observation are inappropriate to the immediate problems faced by resource managers. Palaeontological research has revealed that species have shifted independently over geological time (e.g. Graham and Grimm 1990, Walter and Paterson 1994), and this has been used to support a view that communities are essentially random assemblages of species (e.g. Liss et al. 1986). However, biotic communities with intricate and obligate relationships are recognisable phenomena within our perception 'window', and neither land management nor maintenance of human societies takes place over geological timeframes.

The measures of species assemblages themselves define the 'reference frame' for a community (May 1994). Changes in the community composition, whether in time, space or genetic make-up may be viewed as movement of genotypes through this reference frame. The rate of such movement varies depending on the abiotic and biotic environment and the life span of the individual organisms measured. For example, a community defined using podocarps may indicate stability for hundreds of years, while over this period many bird or insect species may invade or become locally extinct. Small remnants of lowland native bush within grazed areas provide particularly poignant examples of individuals persisting when most of the associated community has become locally extinct, and grazing prevents regeneration of the system. We have only limited knowledge of the extent to which associated biota such as endemic insects and soil microbial fauna remain within such stands, or the degree to which persistence of these indigenous vegetation remnants may depend upon this.

The palaeontological evidence led Walter and Paterson (1994) to suggest that the concept of 'communities' should be removed from the language of ecological theory. This appears strange to those of us who have no trouble recognising mountain beech, tawa, pine or pastoral systems. However, the point of Walters and Patterson (1994) was to draw attention to the false underlying premises of much of the theoretical ecological literature. Models are often

ultimately based on putative interspecific competition within restricted trophic levels, rather than on the inherent suitability of organisms for a local environment. This has led to searches for ecological principles based on numerical combinations of community components, rather than on understanding derived from the qualities of the component species themselves. Williams (1986) pointed out the fallacies of such assumptions in his critique of island biogeography theory. Many comparisons of real systems have revealed the inadequacy of quantitative information without the qualitative knowledge associated with the organisms concerned (e.g. Hurlbert 1984, Samways 1984, Wiegert 1988, Hutcheson 1990, Tokeshi 1993, Holbrook et al. 1994, Tonhaska 1994). More recent evidence (Beck 1997, Grime 1997, Wardle et al. in press) has led to a questioning of broad ecological extrapolations from limited studies in managed production systems.

There are several corollaries to a refocusing on the ecological suitability of component species for a habitat. For example, components of these assemblages are transient at a range of spatial and temporal scales, and therefore there will be an individualistic nature to community composition and interactions within the controlling influences of availability and suitability. This of course has led to discussion of what constitutes a recognisable community (Bastow-Wilson 1993, Bastow-Wilson et al. 1996). Because communities are defined by sampling methodology, and spatial distribution of individuals within particular species may not map closely onto suitable habitat patches (Wiens et al. 1997), single species cannot be expected to indicate all the characteristics of local assemblages. However vegetation communities are recognised by all field workers despite their individualistic natures (e.g. Terborgh et al. 1996), and where sampling spans broad trophic groups, the biology of component species in assemblages can provide interpretation of community processes (Hutcheson 1996).

2.1.4 Lack of baseline knowledge

Only about 13% of the estimated total of c. 13 million species in the world have been identified to date, and the greatest focus of study has been on the chordates, which comprise a mere 3% of the total. Ecological knowledge of most named species (the majority of which are insects) is limited or non-existent (Watson et al. 1995). Improving our understanding of biodiversity will obviously require not only the gathering of new information, but also doing this in a way that returns the most knowledge on how to sustain this resource for the information gathered.

There is an obvious inherent philosophical difficulty in the circular process of selecting indicators of system qualities, which are needed because we can never know the entire biota of systems of any size (Disney 1986a). In addition, it is intuitively obvious that many taxonomic groups are functionally distinct, and adapted for different environments (e.g. frogs and cacti) and will not show similar species richness patterns across a range of localities. Hence, we cannot expect direct parallels between all groups in terms of richness (Curnutt et al. 1994, Danks 1994), endemism (Peterson et al. 1993) or rarity (Prendergast et al. 1993).

Vegetation systems themselves provide the most visible, stable and recognisable indicators of biotic communities, while invertebrates provide the greatest portion of genetic variation or biodiversity. Better understanding of how biological diversity, as displayed by invertebrate communities, is related to vegetation systems and their variation, may provide the means to retain a large proportion of endemic genetic variety through management at the system level. However, the field is so great, the numbers of workers in the more difficult areas so few, and the needs of management so pressing, that we must seek pragmatic approaches to the problem (Erhlich 1992). This calls for parts of the biota to be used as 'indicators' of broader qualities of systems. Moreover, such indicators must be related both to useable entities (i.e. vegetation systems as perceived by managers), and to the heart of biodiversity (i.e. the insect communities).

2.1.5 The need for a systems-based approach

The above discussions suggest the need for collation and comparisons across spatial, temporal and taxonomic scales. The immense range and fluctuating nature of biodiversity means that standardisation must be of method rather than of specific organisms. Biotic interactions, and the consequent multi-factorial nature of individual species response, mean that documentation must be of assemblages. Perceptions and management of the environment occurs at the level of vegetation systems (or communities), and therefore assemblages documented should be related to vegetation systems. Three levels of scale appear useful: level 1, regional/landscape; level 2, local rapid documentation; and level 3, local detailed study.

2.2 CURRENT APPROACHES TO THE SELECTION AND USE OF INDICATOR SPECIES

This section describes a selection of approaches and suggestions for the selection and use of indicator species. To a large extent, they represent studies at the level 3 noted above. They are largely derived from the perspective of single-species plant monitoring, although many authors have acknowledged tacitly or openly the need for broad comparisons of systems, and for the integration of studies.

2.2.1 The limits of change approach

Some of the best examples of the use of *ecological indicators* come from their utility in social issues such as the problem of waste and resource-based recreation, especially where they are part of a 'limits of change approach' (Hedley pers. comm. 1994). This is a framework for establishing acceptable and appropriate resource and social conditions in recreation settings. Conceptually, this evolved from research into the carrying capacity of particular target environments, with selection of clearly defined objectives, establishment of an environmental baseline, and evaluating and monitoring environmental change over time. The process requires managers to identify where, and to what extent, varying degrees of change are appropriate and acceptable. It also involves a shift in attention to include the social conditions in an area. Major stages in the

process are listed here in a resource use context. These can be modified for general ecosystem use:

- identification of issues and concerns within the study area;
- definition and description of the range of development and use opportunities offered by the study area;
- selection of sample indicators of resource use and present environmental state;
- initial inventory of sample indicators;
- specification of standards for resource use and (social) environmental interaction;
- identification of alternative opportunities;
- identification of management actions;
- evaluation and selection of preferred alternatives for management and resource use;
- implementation actions and the ongoing monitoring of indicators.

The limits of change approach was primarily developed for recreation planning, but its general principles have application to a wide range of development, preservation and resource use issues, as well as for the study and assessment of dynamic processes in both natural and modified ecosystems.

2.2.2 Characteristics of indicator species

What constitutes a 'good' indicator species? Indicators do not have to be biological species (although they often are), but can also be abiotic in nature, such as measures of soil texture or water chemistry. However, it is the effects on the biota that concerns us. While abiotic indicators may be easily measured, a relationship between any putative abiotic indicator, and the biota must first be generated across a complete range of interactions and contexts. Therefore it would appear more sensible to develop standard methods to measure the biota directly.

A number of general attributes have been suggested for indicator species. These include:

- being sensitive to change over a short time-frame within the period of measurement;
- being widespread through the study area;
- being widespread through time and not just ephemeral;
- having occurrence related to dynamic processes or functions;
- being clearly measurable, e.g., in size, abundance, growth, structure or frequency;
- being cross-referable to other indicators, while having independent attributes.

GEOGRAPHIC RANGE	LARGE		SMALL	
	WIDE	NARROW	WIDE	NARROW
LOCAL POPULATION SIZE LARGE, dominant somewhere	1 Locally abundant over a large range in several habitats Common spp.	2 Locally abundant over a large range in a specific habitat Predictable spp.	3 Locally abundant in several habitats but restricted geographically Unlikely spp.	4 Locally abundant in a specific habitat but restricted geographically Endemic spp.
LOCAL POPULATION SIZE SMALL, non-dominant	5 Constantly sparse over a large range and in several habitat Sparse spp.	6 Constantly sparse in a specific habitat but over a large range Predictable spp.	7 Constantly sparse and geographically restricted to several habitats Non-existent spp.?	8 Constantly sparse and geographically restricted to a specific habitat Endemic spp.

The classic typology of rare species developed by Rabinowitz (1981), above, provides a useful framework for starting to assess what species may be most appropriate as indicators of ecosystem and habitat health, although we are not aware of this typology being used for the purpose of developing indicator species. What Rabinowitz does is consider geographic range, abundance within that range and habitat fidelity as three components of an eight-celled system.

Two cautions are acknowledged by Rabinowitz:

- (1) that it is a scheme to aid thinking and not a rigid classification, and
- (2) that it is a typology of results rather than mechanisms or causes.

Of particular interest in terms of ‘indicator’ value are those species which are classed as ‘predictable’ especially those which are present at least at some sites in reasonably large numbers. Such species are both habitat-specific, i.e. have a well-defined and narrow ecological niche, while being both abundant and widespread enough to be capable of being readily detected. In the table above, the group felt to be most valuable is in cell 2.

Other useful species are found in cell 4, especially where indicator studies are geographically restricted—endemic species which are at least locally abundant in a specific habitat but restricted geographically. Such species are useful as indicators on a more restricted geographic scale but their usefulness may be extended by use of analogue species which occupy the same functional guild in other areas. Less useful but sometimes appropriate are species which are in cells 1, 3 and 6. These species are locally abundant (and therefore may be readily surveyed) but have larger niche spaces and hence are not always specific to particular habitats, or are species which occur over a large geographic area within a very narrowly defined habitat but are usually locally sparse.

There has been a tendency to use rare or threatened species as indicators. Some of these may be good indicators, at least for special situations, but it is often better to use species that are more widespread, while still being restricted to a very particular range of habitats, i.e. cell 2 above.

Stork and Samways (1995) suggest the following as being useful attributes for indicator species which are applicable to a wide range of organisms in a variety of ecosystems (modified from Brown 1991):

- high taxonomic and ecological diversity (many species in each locale or system);
- close association with and identification of the conditions and responses of other species;
- high ecological fidelity;
- relatively high abundance and damped function (i.e., they are always present and are easy to locate in the field);
- narrow endemism or, if widespread, well differentiated (either locally or regionally);
- well known taxonomy and easy identification;
- good background information (e.g. on genetics, behaviour, biochemistry, ecology, biogeography);
- large random samples, encompassing all species variation, are possible;
- functional importance within the ecosystem is understood;
- predictable, rapid, sensitive, analysable and linear response to disturbance.

A broad-scale grouping of indicator types which may be recognised was provided by Spellerberg (1992). These include:

- *sentinels*: sensitive species introduced into atypical conditions as early-warning devices (e.g. canaries in coal mines);
- *detectors*: species occurring naturally in the area of interest which may show measurable responses to environmental change (e.g., changes in age-class, decrease in population size, changes in behaviour)—heathlands are indicators of nutrient-poor acid soils and some vascular plant species may be used as indicators of woodland type;
- *exploiters*: species whose presence indicates the probability of disturbance or pollution (e.g. in aquatic habitats, the presence of many tubificid worms and/or red chironomid larvae indicates polluted conditions);
- *accumulators*: species which accumulate (bioaccumulation) chemicals in their tissues (lichens, woodlice);
- *bioassay organisms*: selected organisms sometimes used as laboratory reagents to detect the presence and/or concentration of pollutants (some species of trout make suitable bioassay organisms).

In summary, it is desirable that biological indicator species be sensitive and respond readily to change over time, space and habitat, in terms of abundance and replication.

2.2.3 Baseline studies and monitoring

Baseline studies are important as inventory studies that establish a point in space and time against which other points in space and time can be compared and contrasted. The first of a series of monitoring sessions forms such a baseline. Baseline inventories must be: systematic, time and space related and scientifically defensible. The kinds of data necessary for baseline studies is essentially that required for monitoring. Hence, discussion of indicators refers to both base-line and monitoring needs.

Initial emphasis of baseline studies has included:

- inventory of the overall patterns, relative areas, and functional features of the study area(s) in terms of landscape, soils, human influence and essential processes;
- determination within the context being examined of what elements give particular character, and of the social perception of these elements;
- identification of *zones of conflict* which lie between areas of greatly differing character, or where there is potential to adversely affect an adjacent area;
- gap analysis of protected or *nature dominated* sites, areas and systems to identify real missing parts of the system which the region cannot afford to lose.

2.2.4 The need for integrated monitoring

Monitoring is the process of repeated observations, for defined purposes, of one or more elements of the environment according to prearranged schedules in time and space using comparable methods for environmental sensing and data collection. Its primary purpose, as distinct from baseline studies (or inventory), is to document direction, rate and intensity of change (or conversely, of stability).

Most traditional methods of environmental management tend to focus on one or very few environmental parameters at any one time. Ideally a system needs to be developed, of indicators that reflect the state and management of resources or key environmental sectors in relation to long-term sustainability.

3. Vegetation

3.1 INTRODUCTION

The general aims of vegetation analysis can be generalised into three nested levels (Causton 1988):

- large-scale vegetation survey, usually of a new area, for description and mapping;
- small-scale survey of a restricted area, containing different vegetation types, where the objective could either be mapping or ecological purposes;
- more detailed work which might involve comparisons between the different vegetation types or more detailed work on individual species found in the whole area.

Information from each of these levels has potential indicator value although the kinds of questions which may be answered will be different in each case. Repeated large-scale surveys can provide valuable information on natural changes in landscape structure, the effect of changes in watershed hydrology and the effect of human occupation of a region. Information on e.g. fragmentation of natural ecosystems is obtained at the first and second level of scale, while study of individual species at the third level of scale gives more refined information on specific ecosystem components and processes.

3.2 AN HIERARCHICAL APPROACH

Menges and Gordon (1996) use a three-level, hierarchical approach of increasing intensity for the study of rare plants where:

- level 1: focuses on species occurrence by mapping distribution of species and identifying the presence/absence or spatial extent of each population;
- level 2: allows the analysis of population trends and hypothesising about demographic mechanisms;
- level 3: is the most rigorous and involves demographic monitoring of marked individuals, thereby permitting quantitative assessment of demographic parameters such as survivorship, growth and fecundity. Such data can be used for modelling and population viability analysis.

This hierarchical approach can readily be applied to most ecological studies, including not only documentation of rare plants, but also of systems and the processes occurring within them.

The three levels of monitoring as expressed by Menges and Gordon (1996) are shown in the following table:

LEVEL	GOALS	QUANTITATIVE APPROACHES
1. Distribution of populations	Measure trends across populations; hypothesise trends in population size	Descriptive, geostatistics
2. Population size/condition	Measure trends within populations; hypothesise mechanisms	Trend analysis Metapopulation analysis
3. Demographic modelling (of individuals)	Anticipate population trends; understand mechanisms	Population viability analysis

Menges and Gordon point out that it is impractical to intensively study all species especially when time and resources are limited. Therefore they regard their three levels of monitoring as a way of putting resources and time into those monitoring exercises which will yield the best value. The point that these authors make is that these levels can be nested so that, for example, a few populations of a species receive intensive level 3 monitoring while all populations are subjected to level 1 monitoring. This makes sensible use of limited resources. Level 1 is good for initial studies of ecosystem condition or the health of particular populations, and for selection of potential indicator species; but level 3 will yield the most information.

This approach has also been found to be very useful, if not requisite for investigations of indigenous forest health problems in New Zealand. Traditionally, on-site observers such as foresters have recorded insects causing major problems within our forests. A causal relationship has consequently often been ascribed. Identification has usually shown the insect to be endemic and to have shared a long evolutionary history with the plant in question. This raises questions of why the plant species is still extant and why the particular damage is occurring at that particular time and place. In many instances, investigation has shown the insect to be a contributing agent in a process involving many factors. The hierarchical approach has assisted such conclusions in forest health studies on beech forest (Appendix 1.1) and pohutukawa (Appendix 1.2).

In the latter example (Forest Research Institute 1989, Hosking and Hutcheson 1993), level 1 investigation showed the loss of previous pohutukawa cover due to clearance for farmland. The level 2 investigations showed a number of putative causes for health decline of extant trees, but also revealed the lack of regeneration over most of the trees' range due to access by domestic stock. The level 3 study was able to attribute cause of tree decline to possums rather than insects, and to define why and when this occurred. Because of the approach taken, management action to conserve this icon of the New Zealand coastline was directed at the understanding drawn from all three levels, and led to the very successful Project Crimson. An immediate intensive level 3 investigation might have directed action to saving trees, while not halting the loss of the future forest, while use of the level 1 study alone might have directed managers into replanting, but would not have helped the survival of the plantings, or the remnant trees. The hierarchical approach has therefore been found to be capable of delivering knowledge of more use to resource managers, than isolated study at any particular level or site.

3.3 RAPID STANDARDISED DOCUMENTATION — LEVEL 2 STUDIES

Level 2 studies are designed to be reasonably rapid, and as for level 1 studies, standardised methods are necessary in order for data to be cumulative and comparable. Although in many studies, level 2 is designed to provide information on the demography of a particular species, it may also be used to deliver information on plant assemblages (i.e. vegetation communities or systems). In carrying out such vegetation analysis which may have indicator application, several distinct approaches can be adopted. Very broadly, one can distinguish plotless and quadrat sampling, with transect sampling as a variant of quadrat sampling.

One of the simplest sampling techniques is what has been termed in New Zealand, the *recce-plot procedure* (Allen and McLennan 1983, Allen 1992), in which vegetation of a site is recorded from within an undefined area. Essentially observers record what they see at a site, giving cover classes of species within fixed height tiers. Data provide broad information on species components, relative density, vegetation structure and species demography. Recce plots are useful for broad comparison over time and space, and approximate the alpha-diversity of a site. This is essentially the *relevé* sampling of phytosociological analysis. Traditionally, recce plots are limited to what is estimated by the observer to be a homogeneous vegetation type, and in practice when working in forest or scrub, the diameter of plots usually approximates the canopy height of the vegetation. Plotless surveys of this kind have been used for Forest Service Surveys, some Protected Natural Area surveys, parts of the vegetation survey of Mount Cook National Park, Fiordland rapid surveys and many other areas (e.g. Leathwick 1987). Consequently a large amount of historical information exists in this form for New Zealand.

Recce plot sampling is relatively rapid and allows identification of broad vegetation types and some broad parameters within vegetation types. It also allows description of vegetation such as forest with dense undergrowth, which can be difficult to sample in other ways. It is particularly useful for rapid description where the main interest is in major change with time, and allows other parameters to be assessed and associated with a site. The application of multivariate analysis techniques to such data have provided good interpretation of vegetation relationships to underlying micro-climatic and edaphic gradients (e.g. Leathwick 1987).

Although more time-consuming approaches may appear superior, they are not necessarily so, particularly when the live vegetation alone is not the total focus of the study (see discussion in Hutcheson 1996). Spurr and Warburton (1991) compared values for a range of plant species determined from stem density, basal area, vegetation surface area (derived from point intercept method) and vegetation cover (derived from recce plots) as ways of documenting vegetation composition and structure. Their results showed that the different methods emphasised the different aspects measured. For example, basal area emphasised species with large stems such as rimu, while stem counts favoured species with multiple stems such as kamahi. In their study, results from vegetation cover

followed similar trends to those gained from the much more labour-intensive method for describing surface area.

Several plotless sampling methods are based on the idea of measuring distance from randomly chosen points in the study area to certain individual plants, an approach designed for standard statistical approaches, which assume individual plants and sample points are spatially distributed at random. This is unlikely for single plant species, although distribution usually approaches randomness as the individuals of more and more species are aggregated regardless of species (Causton 1988). For this reason at each site the density of all individuals may be obtained first, and then the density estimate of each species calculated in proportion to their frequency.

An alternative is quadrat sampling in which a more detailed record is made within a specified area or plot, and selection of species or all individuals or patches of a species are counted. A standard approach in forestry plots uses a set radius from a central stem, allowing tree density, size and/or age distribution, and individual tree health to be assessed. Such plots may be fixed (i.e. relocatable, e.g. Hosking and Hutcheson 1999) or non-fixed (e.g. Hosking and Hutcheson 1988).

Quadrat area is generally made appropriate to the vegetation type being sampled and to the questions being asked, and a series of nested quadrats of increasing size is often established before a survey is started. The relationship between increasing plot size and increasing species number is plotted, and the change in slope with decreasing rate of accession of new species (or asymptote) is traditionally used as a measure of ideal plot size. There are obvious problems with this approach where comparison is required of changes in vegetation succession over time. For example pure stands of early succession manuka would require smaller plots than the more diverse and taller vegetation appearing later in the succession, leading to difficulties of comparison if smaller, manuka-based, plots are used. This is a problem with the traditional criteria for recce plots also.

Utility for the purpose drives all methodology. At the second level of hierarchical inquiry, the purpose is the rapid (i.e. cheap) gathering of data for the comparison of systems in time and space; hence standardisation becomes the major requirement.

The traditional use for recce plots required them to be representative of vegetation types, and this led to them being placed in homogenous vegetation only. However the current need is for indication of change also, and change is often associated with system boundaries. A simple way of overcoming these problems for level 2 studies is to allow recce plots to be placed in any location, and to designate the area covered as being approximately that of forest plots, say 30 m diameter. Plots may also be made relocatable with a central peg if required. This would satisfy the requirement for standardisation over broad areas and time scales.

A further question concerns whether to sample in a random or regular manner. Whether one uses one or the other is usually determined by the purpose of survey. If the primary purpose is vegetation mapping and description, regular sampling may be preferred. But if the primary purpose is ecological and the

detection and assessment of correlations between species distributions and environmental parameters using an inferential statistical approach, then random sampling has been the traditional approach. Random siting of sample points is time consuming; an alternative which retains statistical validity is to site a transect or sampling line randomly in relation to general environmental parameters and then position plots at regular intervals along this line.

It is worth noting that there are no true treatments in descriptive ecology, in contrast to experimental ecology—only the subjective stratification made by the ecologist. Further discussion of the appropriateness of the traditional statistical methods in an ecological context and of multivariate approaches is given in section 5.6.

3.4 INDICATOR VALUE OF SPECIES RICHNESS

Botanists, and the managers who use their work, usually assume that species richness (number of species in a unit area) is the parameter likely to be a dominating component of biodiversity assessment, and also that species richness will tell us most about the condition and trends in vegetation. Species richness *per se* does not necessarily confer high conservation value. Take the example of two sites which each have the same number of plant species, but one of which has an exotic component of 80% whereas the other has 95% indigenous species. The likelihood is that the first site is profoundly disturbed and possibly in a transition to a very different biological community. A second example is where there is a predominance of indigenous species (say both at the 90% level) but in one instance there has been loss of most woody species through storm and in the other loss of most herbaceous ground cover through grazing. Again species numbers and the percentage indigenous component may be the same but 'condition' or 'health' dynamics and prognosis for the community may be very different. A third example is a comparison between two endemic systems, where one is relatively species-rich, but extensive, while the other is species-poor, but limited in extent. Conservation value is recognised as being higher in the more limited entity. These examples highlight the fact that it is the quality of the community components, rather than simply the quantity, which is the feature required by both the ecologist and the resource manager.

A further problem in concentrating on species richness is that of uncertainty. For large areas it is generally not practicable to develop complete species lists, so there is a need to settle for estimates rather than exact counts of biodiversity in terms of species numbers (Palmer 1995). As discussed in section 2.1 and Palmer (1995), floristic data are largely determined by environment and vary at every spatial scale from a few centimetres to many kilometers. Floristic data exhibit several properties shared by many other (if not all) biotic groups. They have spatial dependence, which means that as sites become further apart they become less similar in their species content. They are fractal in nature, which means that distributions have patterns across a wide range of spatial scales and can be subjected to fractal analysis. They adhere to the 'law of infrequency' (i.e. the majority of species are found in the minority of locations). They have asymmetrical errors (errors in local lists are skewed towards incompleteness).

They are also variable in quality and, as plant species are themselves hypotheses, taxonomic investigation will mean inevitable changes in species concepts and names.

As discussed in section 3.3, there is a traditional tendency for botanists to set up plots in areas of 'representative homogeneous vegetation' and indeed this is often assumed with plotless sampling and is a tenet of Protected Natural Area (PNA) sampling. However, a more dynamic view of ecosystems (Fiedler and Jain 1992, Attiwill 1994, Reice 1994) indicates that transition zones, boundaries and ecotones are often sites of species richness but these tend to be under-sampled. Unusual features such as tracks, streams and boundaries must not be excluded from sampling even though they may be heterogeneous at a micro-scale. Many species are associated with, and provide indications of, system boundaries.

3.5 LEVEL 3 STUDIES

Despite the uncertainties noted above, comparable methodologies at defined scales of investigation allow researchers to reach valid conclusions of influences within natural communities. Palmer (1995) gives a detailed discussion of techniques, limitations and issues involved with this. Palmer notes that we still do not have the statistical procedures to give entire satisfaction for the problems posed by estimates of species richness. As discussed in section 3.4, species richness may not be the appropriate question, rather it is the qualities of the entities present, and their functional interactions which best characterise the system and the processes occurring.

The application of multivariate techniques can be of major advantage in this area. These do not depend on the researcher defining the 'treatments' within and between which the variation is evaluated by statistical inference, but simply display objectively summarised groupings along indirect gradients, allowing the 'data to tell the story' (see section 5.6).

3.5.1 Demographic monitoring

In monitoring indicator species there is often a tendency to simply measure presence and absence, to count numbers of individuals or to estimate density or percentage cover. However, demographic monitoring is both relatively easy to do and yields valuable information about the 'health' and fecundity of populations that can then be correlated with environmental changes. Indeed demographic monitoring has been described as an essential component of effective conservation management especially for rare plants (Owen and Rosentreter 1992).

In a review of the available techniques, Owen and Rosentreter (1992) note that traditional demographic techniques have inherent statistical and logistic problems which may mean that they do not provide a clear view of population processes, and hence may fail to realise their potential as indicators. They suggest that because most statistical procedures require unbiased data, there should be some randomisation of sampling unless one can specifically select a statistical method that is reasonably forgiving. Perhaps the simplest device is a grid superimposed on a study population or area; this can utilise a spreadsheet

such as Lotus 1-2-3 or Excel. It has the advantage that one can not only use it to randomly select study cells (e.g. by using a random number generator), but can also use the same spreadsheet for entering data. Another way of achieving randomness is to use 'belt transects' in which a number of plots are located at regular intervals along a transect line, but where the transects themselves are located on a random basis.

Once the sampling system is established the question arises of what to measure. For perennials, size-class and life stage categorisation are generally accepted as desirable. Age classes can be determined without knowing the absolute ages represented, by assigning classes on the basis of morphology. For instance, classes could be determined on the basis of the number of branches or flowering stems, diameter of the whole plant or major stem at breast height or estimated plant volume. Age classes that are sufficient for indicative monitoring can usually be established without having to have detailed knowledge of life histories.

Owen and Rosentreter (1992) note that, 'a common problem arises when managers wish to compare the distribution of size classes between populations, because plant size at any one location may be determined largely by local environmental conditions. Size classification schemes should address the problem by defining size classes that span the species' entire range of sizes, even though each population may not contain plants in all size classes'. An alternate approach which is often appropriate for New Zealand trees, is the use of growth phases. Older trees of e.g. pohutukawa often fall, reshoot and reenter a vigorous juvenile growth phase that may be more resistant to health problems (Hosking and Hutcheson 1993).

Reproductive output is obviously important and can be a very sensitive indicator of changing environmental conditions. Lack of flowers on pohutukawa may be indicative of a redirection of resources into foliage growth after bud browse by possums (Hosking and Hutcheson 1993). Seed set per fruit is a useful measure, especially if it can then be multiplied by number of fruit (sampled for part of a number of plants—e.g., fruit on a randomly selected lateral branch of every fifth plant). This is readily done where there are few seeds in each fruit, but where fruit have many seeds or seed number is highly variable, it may require counts from a large number of plants. A serious attempt to monitor reproductive effort will require actual determinations of seeds per flower and fruit, flowers per inflorescence, and the variance associated with those estimates, sampled over a period of several years. However, for highly sensitive indicator species where higher-level study has led to a belief that reproductive effort responds readily to environmental change, such intense sampling of a single species may be extremely productive.

For information on the actual mechanisms leading to particular levels of seed set, more intensive procedures designed to estimate levels of in- and out-breeding, and possibly genetic techniques may be necessary. A readily measured parameter is seedling establishment, especially if it is associated with germination trials associated with the seed bank in the soil (this is the store of natural viable seed built up in the soil adjacent to the target species).

Obstacles noted by Owens and Rosentreter (1992) include missing data, often resulting from not being able to get to the study site or from weather which

precludes data collection. There are statistical procedures to cope with this. It may be difficult to randomise data collection because of land tenure problems, small or unstable populations, or the ephemeral nature of a habitat. Catastrophic events can totally destroy a sampling site so that critical indicator sites should be replicated.

3.5.2 Resampling frequency

Discussion with a number of field ecologists indicates that two particular problems which arise with indicator programmes and monitoring are:

- (1) how frequently should sites be re-sampled, and
- (2) how intensively should sites be studied or measured.

There is no easy answer to the first, and in fact there appears to be virtually nothing in the literature relating to establishment of rules for frequency of remeasurement. The general comment was that frequency of remeasurement tended to be intuitive, 'seat of pants', and opportunistic. The following have been suggested as guidelines:

Once plots, recce-type sites or transects have been established, initially remeasure more often than you think necessary and then, if appropriate, measure at longer intervals of time rather than the other way round.

If the total time from initiation of a perturbation to the point where equilibrium is achieved is t then the interval between remeasuring should be no more than $t/10$. In other words if the recovery time from a perturbation is 20 years then remeasuring every 2 years would probably be appropriate. Hosking and Hutcheson (1999) remeasured every 2 years when monitoring effects of beech forest blowdown. This showed that the event itself lasted over a decade, with remnant trees still dying 10 years after the storm, so estimating t may be problematic.

Sites should, as far as possible, be remeasured at the same time of year or in the same season.

Start with more plots or transects than you think you need and reduce the number on the basis of initial results—it is often possible to prioritise plots and transects—examining a subset more frequently than others. However, if a core group is selected for long-term study then it is recommended that you do not switch to alternative sites later (unless one or more sites are completely destroyed).

With respect to intensity of measurement there is a feeling that it is better to measure too many parameters and then reduce the number, rather than to find that there were essentials which should have been measured but were not. Unfortunately such an approach may be very time-consuming (and therefore expensive). The suggested hierarchical approach that introduces this section can give very good guidance to this general problem. It allows a better problem definition using limited resources, while retaining a flexibility to heuristically improve study direction. This enables a better definition of both the questions being asked, and the best approaches to answering them.

A tool which can be extremely useful at a preliminary level for plants and which has been used in draft recovery planning for individual species as well as for

assessing vulnerability of individual vegetation stands, e.g. on the Chatham Islands, is a quick measure of 'vulnerability index'. This also has potential for helping select species that may be good indicators. It is a rapid assessment of the sensitivity or vulnerability of different biological attributes of a species. An example of this is given in Appendix 2.

In practice, use of vegetation and plant indicators—especially for a regional programme—will often be at several levels: vegetation and landscape, population, and individual plant. A necessary relationship exists between trends that drive the management of ecosystems, species and populations. Species management comes down to management of particular populations at specific sites. But species conservation cannot be considered independent of the habitats, ecosystems and landscapes where they occur. The use of land adjacent to the target site is also relevant to conservation of particular populations.

At the individual species level most approaches will be either demographic monitoring or ecophysiological monitoring which concentrates on genetics and resource allocation (Davy and Jefferies 1981). A large number of techniques which examine demographic, genetic, and physiological characteristics of populations are available for adaptation to monitoring programmes (e.g. Wells 1967, 1981, Menges 1986, Pavlik 1987, Pavlik & Barbour 1988).

Of primary importance to a demographic monitoring programme are relatively intensive (i.e. level 3) studies that document population trends (stability, growth, or decline) at a specific site. Practically speaking, 'intensive' may mean 2-3 years of following marked plants in permanent plots. These studies may have to be extended for longer periods in order to provide meaningful data (B.M. Pavlik pers. comm.).

The more detailed programmes, which may only be able to be used in a relatively small number of selected situations, monitor individual plants. This may mean frequent measuring or mapping of plants (perhaps every 2-3 weeks) in order to achieve this resolution, so it is not surprising that these studies are rarely sustained for more than a few years. In the most detailed studies, when there is a quite critical situation to be assessed, the physiological status and reproductive performance of selected individual plants becomes the primary focus. This has been found to be particularly true for forest health studies in New Zealand.

With such intensity of effort, the parameters measured and scale of measurement must be such that they will indicate what manipulations, sometimes small-scale, would relieve undesirable stresses. If the measurements are associated with manipulation of plants (e.g., inducing drought stress) they should be able to indicate the effectiveness of manipulation, or what other manipulation may be appropriate. Some techniques such as ecophysiological measurements based upon the carbon balance approach can sometimes give immediate information on the effects of small-scale, experimental treatments without sacrificing individual plants. Time is an important element—for precarious populations and rapidly changing situations there may be no second chance if the wrong things are measured or if monitoring is delayed. For these reasons it is imperative that good advice is obtained regarding the precise technique, treatment and analysis most appropriate to level 3 studies.

Although not yet published, a pertinent New Zealand example of how intense monitoring can help in both management of a target species and in indicating long-term trends in local environment, is provided by *Helicbrysum dimorphum* (B.M. Pavlik and D.R. Given unpublished data), a very localised New Zealand liane of the Canterbury mountains. Only two significant populations now exist. Mapping of sites and estimation of population size and flowering (level 1 and 2 observations) was useful but did not really provide detailed-enough information on either habitat or species dynamics. However, two-seasons data on detailed ecophysiology (especially seasonal growth rates and moisture stress using data from selected individual plants at two sites) suggest that the species is highly sensitive to late summer drought and has an exceedingly narrow niche requirement. At a local level it has considerable potential as an indicator of change in vegetation structure, hydrology and climate, and in addition this level 3 investigation has yielded predictive information for re-establishment of the species.

Other potential indicator plants with physiological value especially in drier parts of New Zealand almost certainly exist. Although the techniques for examination require specialised equipment and knowledge of techniques, once identified, needs can generally be supplied by university physiology laboratories, and by some Crown Research Institutes.

The following guidelines have general application to monitoring of plant indicator species (based on Davy & Jefferies 1981; Given 1994; Palmer 1987; Wells 1981):

- (1) It is essential to know precisely what sort of results are being sought and what the results mean.
- (2) Conservation of populations requires an understanding of the dynamic aspects of the population; the most important are recruitment (births), mortalities (deaths), survivorship, longevity and turn-over of the population.
- (3) If monitoring at the same time each year, only one stage in the life-cycle (in annuals) or one phenological stage (in perennials) may be considered. It is often necessary to study populations several times during a 12-month cycle.
- (4) Careful monitoring of populations on a regular, often annual basis, using a technique which enables the same individuals to be recorded and examined each year, will provide basic facts about the population from which predictions can be made about future structure.
- (5) Attention needs to be given to flowering and fruiting, and to reproduction of both above- and below-ground parts of plants. This should be integrated with environmental factors and various management regimes.
- (6) Many plants have particular biotic relationships with other organisms. As an example, mycorrhizal associates of plants such as orchids, and saprophytes are important but are often imperfectly understood, and special techniques have to be developed before progress can be made in this field.
- (7) Implicit in intensive ecophysiological monitoring is the idea of examining responses to environmental disturbance. Disturbance may correspond with a potential management procedure such as grazing, nutrient addition, irrigation, drainage or burning (although this should not be assumed).

(8) Variables such as plant size and seedling data should be included in studies. Inferences can be based on carefully selected random subsamples reinforced by use of controls. This may be difficult if the population is small. Under field conditions, it is not always possible to randomise or to have independent replications.

(9) A critical review process should assess goals, sampling methods, and experimental design of proposed projects. If a population continues to decline despite intensive monitoring and small-scale manipulation, more drastic steps including emergency greenhouse propagation may be necessary.

Underground parts of plants, and interactions between the root systems of different plants should not be ignored; although usually hidden from view, underground organs are a vital component of plant biology, and of vegetation (St John 1987; Stanton 1988).

A further set of key points for level 3 studies are suggested in a review of vegetation monitoring (Stewart et al. 1989):

- the need for clearly defined and appropriate objectives;
- the use of methods appropriate to the objectives and the type of vegetation involved;
- the need for careful site selection and an appropriate sampling strategy;
- the importance of adequate field marking for relocation of plots and transects, and the need for periodic checking of plot markers and plant tags;
- the detailed recording of location, objectives, methods, and recording forms so that re-surveys are possible, even many years later;
- choice of an appropriate monitoring interval, normally dictated by the rate of change, the detail required, and the resources available. The most effective monitoring interval may not be apparent until after initial re-surveys, and fluctuating rates of vegetation change may prompt changes in the monitoring frequency at any stage;
- the use of well trained and dedicated staff capable of careful and often tedious field measurements and data interpretation;
- care to minimise the physical impacts of monitoring.

Exotic species should not be overlooked as indicators. Weedy species are often very good indicators of land degradation although the precise nature of the degradation may not be fully known. Good examples include *Hieracium* (overall presence, percentage cover, species make-up and rate of spread), *Rosa rubiginosa* (sweet brier) and *Ulex europeus* (gorse). A common index in rapid assessment of vegetation remnants is an estimate of the percentage of ground occupied by exotic species. A five-point scale, for instance, ranging from insignificant to 'dominated by exotic species' can be used to provide an initial assessment at the level 2 scale in the hierarchical model. These can be followed up later by (level 3 studies) giving more sophisticated estimates of cover and exotic species composition.

3.5.3 Non-flowering plants

Relatively little use has been made to date of non-flowering plants as indicators of environmental change and habitat condition, although many species of lichens, mosses, algae and other non-flowering organisms may have potential. For example, mosses were found to reflect aspect and forest canopy closure by Mathers (1992).

Another example comes from the Antarctic (Markham et al. 1990). This example is unusual insofar as the indicator value of the target species was realised through another investigation for a rather different purpose. Initially *Bryum argenteum* was collected and analysed for leaf flavonoid chemicals to see how well different moss species could be identified chemically in the absence of sexual reproduction. It then was realised that because flavonoids are sensitive to UV-b radiation, this species might provide a means by which UV-b radiation levels could be monitored in relation to global climate concerns. Because flavonoids are very stable, use could be made of historic records in the form of herbarium specimens, and as a result, a UV-b curve was constructed back to the 1950s, confirming biologically the curves obtained through upper atmospheric physical observations.

The potential value of this species as an indicator has yet to be fully realised. The technique is reasonably easy and cheap to use, and *Bryum argenteum* is both easily recognised and is one of the most widespread of all plant species. This is an interesting example of a species indicator which, although it has an extraordinarily wide niche, has a particular feature (in this instance production of particular chemicals) which allow it to be used to monitor change in a very specific area (in this instance UV-b radiation).

Lichens already provide an excellent example as proven indicators of air pollution, especially in urban areas. Use of both lichen cover on tree trunks and mapping the occurrence of species sensitive to different levels of pollutants can lead to the development of maps with isochors that relate biological patterns to areas of differing pollution intensity. Most tend to be restricted to particular ecological niches. Most lichens respond to the chemical nature and texture of the substrate as well as microclimatic variables. Tree lichens (*Corticolous* sp.) respond to the conditions of the bark rather than to tree species. Changes in pH of the bark can change the lichen community. For example in Britain, birch and pine have more acid barks than oak, and tend to support different communities. The lichens that occupy barkless wood (*Lignicolous* sp.) tend to be species that normally live on acid bark. Rock dwelling lichens (*Saxicolous* sp.) respond in a similar manner to those on trees. This may be observed in cemeteries where headstones may be made from different stone types, e.g. limestone or granite display different lichen growths. The ground lichens (*Terricolous* sp.) can form dense swards at high latitudes given favourable conditions, e.g. tundra 'lichen-heaths' in boreal and subarctic regions. Some lichens are cosmopolitan e.g. *Parmelia sulcata*, but most tend to be restricted within major vegetational zones and subzones. Relatively few species are endemic to small geographical regions. In Britain only two or three species are truly endemic.

Few examples exist of such lichen studies in New Zealand, although they can be carried out very economically and can involve numerous observers, thus lending themselves to co-operative urban exercises.

3.6 GENETIC CONCERNS

Genetic analysis in different parts of the world is showing that apparently simple and homogenous populations sometimes consist of numerous smaller, genetically distinct populations (the meta-population concept). This appears to be likely where a population is diffusely distributed over a large area in varying densities, and there are considerable differences in population sizes. Meta-populations have been known for over a decade among animals but are now being recognised as being widespread among biotic groups.

Where there is reasonably good information on animals which have mutualistic relationships with particular plants, this can sometimes be detected by estimate of pollination and fruit dispersal patterns. Use of this may determine 'likely neighbourhood distances' that correspond to the extent of genetic interchange between patches of individuals. An example of this might be to determine the mean and variance of distances to which vectors such as insects, birds and wind transport fruits from source plants. At other times such estimates may be out of the reach of field managers, because only more sophisticated techniques such as isozyme or seed protein study may indicate genetic structure of populations. However, such features as genetic and chemical structure of populations should not be ignored for highly valuable indicator species because changes in these may indicate important long-term environmental changes.

3.7 RECOMMENDATIONS FOR PROGRESS IN NEW ZEALAND

It is essential (and stressed by numerous ecologists) that there be an adequate range of long-term monitoring studies for effective ecological interpretation and management of New Zealand's terrestrial vegetation.

The hierarchical approach of Menges and Gordon (1996), which was also used in beech forest and pohutukawa health studies in New Zealand (Hosking and Hutcheson 1986, 1988, 1993), should be considered as a standard one. This approach allows a better definition of the questions that need to be asked. It thereby assists in the selection of those species, sites or plots which will be examined more intensively. The use of standardised methodology, even if this only occurs at the larger scales (levels 1 and 2), allows for the comparisons over space and time that are crucial for management of resources that are located over widespread regions.

A great deal of level 1 information is currently being acquired through the use of remote sensing. Mapping of vegetation systems from this source is currently being entered into the Geographic Information Systems (GIS) environment by resource managers in conservation, production systems and local authorities. The widespread availability, improving resolution and standardisation of such information, will allow rapid indication and collation of the changing extent, fragmentation, and for some systems, the changing health of New Zealand's natural vegetation systems.

This information may be better defined, where this is deemed necessary, from relatively rapid level 2 studies. An appropriate approach is the use of standardised (forest) sized recce plots, particularly if these are not restricted to being located within homogenous vegetation types.

The majority of level 3 indicator experiments also use some form of plots, which may be ordered (e.g. arranged into transects) and this allows future replication of measurements with some degree of certainty about the validity of comparisons over time. No one approach to level 3 studies will give all the answers, nor is there necessarily any 'right' approach to the use of indicators; but some approaches are more appropriate than others. There is also considerable scope for innovative level 3 studies of plants as indicators of ecological processes, and some of these have been outlined above, or are included in Appendix 1. Even for level 3 studies, however, there needs to be emphasis on using methods that can be standardised, providing consistency in spatial, temporal and taxonomic scale.

There are problems with the application of traditional experimental statistical methods to natural systems. This is because data are necessarily descriptive (there are no treatments as such). While there may be a need to develop a better range of inferential statistical methods for use with natural system data, full use should be made of currently available multivariate tools which have been developed to circumvent these problems. These are powerful mathematical tools which can summarise and present data in an informative manner that is understandable to both the researcher and the resource manager.

Most current indicator studies involving plants tend to focus on vegetation patterns and structure, or on particular plant species, and much of the past and current work is amenable to being collated within the hierarchical model. With minor modifications and additions, current methodologies could be standardised to allow level 1 and 2 collations and comparisons. Such a move would save considerable resources, and would provide resource managers with far more accessible data within the short term.

Level 3 studies consume considerable resources, but with the context and promising directions for studies being derived from levels 1 and 2 in the hierarchical model, a natural ranking in priorities will occur.

4. Vertebrates

4.1 THE UNIQUE NATURE OF THE NEW ZEALAND SITUATION

Terrestrial vertebrates are represented by both indigenous and invasive species within New Zealand's natural estate. The endemic vertebrate fauna consists of birds, bats and reptiles. The kiore, which some consider to be native but which is not endemic, is the only mammal which may possibly be regarded as part of the New Zealand fauna apart from bats.

The ground-dwelling vertebrate mammal community consists almost exclusively of introduced species. Deer, rabbits, pigs, wallabies, possums, rodents, mustelids, and domestic stock, although introduced to New Zealand, have the greatest influence on the condition of our natural estate. As a consequence, our approach to the measure of vertebrate influence on ecosystem condition is skewed in a direction that is almost unique in the world. In other areas of the world, research may be directed at examining the effect of management regimes on the spread into new areas of vertebrates which are endemic to these regions (e.g. voles, shrews and woodmice in Europe). In New Zealand, in contrast, we are generally investigating the resilience of a particular ecosystem to invasiveness, or investigating the habitat quality for particular endemic species, e.g. kiwi, in the presence of invasive species such as possums, mustelids or dogs.

The impact of exotic vertebrate species on New Zealand's natural estate has been unprecedented in the annals of natural history. Their control and the measure of their impacts is of primary importance in the preservation of species and particular habitats in this country. Their presence is an indication of threats not only to the survival of many of our threatened and endangered vertebrate species such as kokako, kiwi, and woodpigeon, but also to our invertebrate species (Rickard 1996). Through prey switching during periods of low populations of vertebrate prey (mustelids), or the requirement for proteins or high-quality food during certain periods of the year (possums) (Owen 1993), these vertebrates can influence community structure in ways that are not readily apparent. The challenge to the stewards of these estates, is how to measure these influences in the most efficient and accurate ways possible, without consuming an inordinate amount of available resources. The use of indicator species, or groups, may be one approach to this problem. The questions here are: what do we measure, where within the ecosystem do we concentrate our studies, and are there adequate resources to carry out these studies without endangering other programmes?

Much is known of the requirements for the survival and health of many of New Zealand's endemic vertebrate fauna. For example, the kokako may have its population locally reduced by the direct influence of predators such as the stoat or the ship rat, or indirectly by the destruction of habitat and food source by

possums. The possum may also opportunistically predate the eggs of kokako. In recent times it has become apparent that the possum, which was once considered a threat to plant species only, is having a direct effect throughout the ecosystems in New Zealand. There is evidence of predation on eggs, chicks and adults of at least six New Zealand bird species (kokako, North Island brown kiwi, harrier hawk, fantail, kereru (native pigeon) and North Island saddleback (Innes 1995). Since Clout's (1977) study of possum ecology in pine forests there is increasing evidence that invertebrates also account for a substantial portion of possum diet (Warburton 1978, Cowan and Moeed 1987, Owen and Norton 1995, Rickard 1996). Indeed, the introduced smaller mammals within our natural estate appear to have effects which may be measured right throughout the ecosystem.

The presence of breeding populations of kokako may be indicative of predator absence and good habitat quality, i.e. low numbers of exotic herbivores. Conversely, possum may be regarded as indicators of threats to certain tree species, and with diet switching, may also influence bird populations (through egg and chick predation) (Innes 1995, Rickard 1996). Invertebrate populations may be influenced directly through consumption by possums, or indirectly by loss of habitat and food source.

The larger mammals such as deer do not directly affect the vertebrate and invertebrate fauna through predation. However, without population control measures, they remove palatable plant species, changing or destroying habitat, and limiting the resources for survival and breeding of many species. Deer selectively predate (browse) certain plants, so their presence indicates threats to palatable species. Lack of palatable species in areas where they have been listed, may indicate large numbers of particular grazing species (e.g. pohutukawa, cabbage trees, ribbonwood by domestic stock). Rabbits may be viewed as indicators of disturbance through overgrazing, or inappropriate cultural use of the land.

Our overriding question is, 'where can we most efficiently measure the influence of particular species?' Exotic vertebrate species within New Zealand are past the point where eradication is feasible. We must look at management of species on a system-wide basis. The depredations wrought by these introductions are not constant across the ecosystem. Kea, tui, bellbird, fantail, and pukeko appear to be coping reasonably well in our changing environment. Sometimes these may require the intervention of conservation managers to maintain breeding populations, and e.g. the influence of mynah birds on our smaller passerines in the North Island is somewhat problematical. Kiwi, kaka, and woodpigeons appear to occupy an intermediate place in terms of resilience. These are species which are sensitive to the disturbances caused by introduced vertebrates but which are patchily successful. They can all be predated upon by mustelids, rats and possums. The kakapo and black robin are examples of species which have been driven to the verge of extinction by anthropogenic activities, particularly the introduction of predators and exotic herbivores. These species are regarded as threatened.

A great deal is already known about the exotic vertebrate fauna in New Zealand (Innes 1990, King 1989, 1994) and their effects on the endemic fauna. Technologies that allow for population and impact studies have been well

developed. The interpretation of these data is usually topic-specific, e.g. what is the population size of stoats in a particular forest and what effect is this having on the endemic bird life? In some cases it may be more beneficial to study the system from a lower trophic level. We know that rats, stoats and possums can have an effect upon several components of an ecosystem. The ship rat is an omnivorous generalist and seeds, fruit, green vegetation, bark and fungi can make up 80% of its diet. They are the most likely of the invasive species to produce irruptive populations (Daniel 1978) and during these times may have an inordinate effect on the ecosystem as a whole. Indeed, Tenebrionidae (darkling beetles) have been found to be predated in large numbers by rats (Watt 1980). Rickard (1996) demonstrated unequivocally that small mammals can have an effect through all trophic levels of the New Zealand ecosystem and these observations are supported by earlier studies, e.g. Owen (1993).

4.2 EXPERIENCE OUTSIDE NEW ZEALAND

Studies of vertebrates outside New Zealand concentrate on species that are usually endemic to the region. Environmental changes can influence the community structure of vertebrates to a high degree (Hanski 1986, Iriarte et al. 1987, Dickman 1988, Walsh 1990). These changes may be due to food resource (Hanski 1990) or habitat complexity (Iriarte et al. 1987) or vegetational composition. Shrews (*Sorex minutus* and *Sorex araneus*) and woodmice (*Apodemus sylvaticus*) prefer drier areas of forest habitat and as the forest matures their numbers become more stable and constant (Walsh 1990). These populations can reach epidemic proportions when food resources become available above normal background levels, e.g. during periods of high insect herbivore numbers (Hanski and Parviainen 1985, Hanski 1990). From this we may deduce that there are multiple indicators within the system. Dry conifer forests indicate that shrews and woodmice are resident; high populations of these small mammals may indicate that there will be an irruptive population of predatory small mammals. Conversely, presence of these small mammals indicates a relatively dry forest habitat, and high numbers indicate that there is a large supply of invertebrate prey in the area. The indicator in this case is dependent upon the question asked.

A small mammal which lives in close proximity with shrews and woodmice is the vole (*Clethrionomys glareolus*). In some areas their habitats overlap. The vole prefers wetter and more open conditions to the other small mammals, and studies have shown that the vegetation within their habitats is significantly different from shrews and woodmice (Walsh 1990). Indeed these differences may be measured in terms of metres. The responses of these mainly herbivorous mammals is to the vegetation, and studies of changes in vegetation community structure may give information on their presence or absence, or of potential population levels. For example, a hole in the forest canopy with a ground flora dominated by *Juncus effusus* and *Ranunculus repens* will be occupied by voles rather than shrews or woodmice (Walsh 1990).

All of the foregoing raises the question: How can vertebrates be used as indicator species?

Vertebrates have particular habitat requirements based upon food availability, seasonality, and habitat heterogeneity. Direct counts of, for example, birds or deer, may not be an indication that they are resident within an area. They may be just tourists. A good example of this may be found in studies of the spotted owl (*Strix occidentalis*) in North America. This species has been the focus of a great deal of controversy in the past ten years. It is often quoted as being an indicator of 'old growth forest'. Studies carried out to test these hypotheses demonstrate that not all available habitats are occupied (Martinez and Jaksic 1996, North and Reynolds 1996, Verner et al. 1992). So, although a resident population of *S. occidentalis* may indicate the presence of particular habitat types and complexity, their absence does not mean that this habitat cannot be found within the area of the study. This may be transferred to the New Zealand situation where habitats exist which could support populations of e.g. the short-tailed bat (*Mystacina tuberculata*) (Lloyd 1994) or kokako.

The New Zealand situation is unusual in many respects, as habitat which is suitable for endemic vertebrates is often not occupied because of introduced predators, rather than the inability of the vertebrates to breed enough offspring or disperse into these areas. Often the limiting factor, excluding habitat destruction, is predation. Conversely the presence of kiwi within a habitat does not mean that introduced predators are absent. The logical conclusion then is that studies of vertebrates within New Zealand must be area- and species-specific, utilising an hierarchical approach to the questions posed. Is the particular species present? If not, are all habitat requirements available e.g. vegetation type, cover, nesting areas, sufficient area for foraging? If these are present we may ask, in the case of kiwi, if the invertebrates upon which they forage are available (young soils may not have a deep enough organic horizon to support a sufficient resident invertebrate community: C. Rickard pers. comm. 1997). In some cases the management of a habitat for vertebrate occupancy may conflict with the objectives of other programmes, e.g. vegetation management. Goshawks and tawny owls in Britain require a patchwork of habitats within a limited area such as shrubland, pole stage trees and mature forest (Petty 1989a, 1989b). In areas where the objective is to return climax forest regions, this would be an unsuitable indicator species. Woodland bats in Britain are considered to be sensitive species and their management requirements are based upon insect prey availability and the structural complexity of vegetation (Mayle 1990). Although the bats are dependent upon the complex woodland habitat for refuge and food source, their foraging areas are in the open over ponds and within forest rides. Again there would be conflicts with other management regimes, such as the preservation of climax forest systems, if bats were to be chosen as an indicator of suitable systems.

Because of their size, adaptability and vagility the use of vertebrates as indicator species is seldom used. Indeed one of the most comprehensive studies of management influences in the Southern Hemisphere 'The Worsley Alumina Project: Flora and Fauna Studies' (Majer 1985b) did not use vertebrates as indicator species. The indicators used in this project were vegetation and invertebrate species. The final success of a particular rehabilitation project was assessed from assemblages of vertebrates with different ecological requirements, which themselves could be measured at a lower trophic level, i.e. via the invertebrates and the vegetation (Majer 1985b).

In many ways New Zealand efforts are peculiar when compared with other countries. We concentrate upon the maintenance, establishment or preservation of species that have been classified as:

- Rare/endangered—e.g. kokako, kaka, red-crowned parakeet, yellow-crowned parakeet, brown kiwi, green spotted kiwi, yellowhead;
- Patchy—e.g. weka, N.Z. Robin, Rifleman, Bellbird, Tui, Kereru;
- Refugee—e.g. saddleback, stitchbird, little spotted kiwi (Clout 1989).

The methods used for these projects are well tested, such as direct counts of birds from calling stations, radio tracking and nest observation. For invasive species, methods include scat counts and vegetation damage assessment (possum), breakback traps and tunnel traps (rats) and baited trapping tunnels (stoats).

4.3 CONCLUSIONS

Vertebrate natural history in New Zealand is biased toward invasive species, and endangered or threatened species behaviour, hence overseas experience in the influence of vertebrates on habitat may not be applicable in New Zealand.

It is suggested for overall management of the conservation estate, when utilising indicator groups, that greater information may be gained from the use of invertebrate and vegetation communities. Much of our vertebrate species assemblages have been decimated and the test of their survival is achieved by direct observation. A knowledge of the underlying community structures may be of more benefit to the maintenance of our endemic vertebrates than simply studying the species of interest.

5. Invertebrates

5.1 INTRODUCTION

As explained in the overview, the use of species assemblages is required for both defining habitats at the scale of management, and for comparing the biodiversity qualities of these habitat units. Vegetation assemblages can characterise habitat types, but the majority of biodiversity is present as invertebrate communities and, most particularly, as insects.

Most invertebrate groups display affinities with particular regions, vegetation systems, edaphic qualities, or anthropogenic treatments in relation to the biogeographic history and ecology of the organisms studied. Examples include molluscs (e.g. Barker and Mayhill in press), nematodes (Yeates 1995, Freckman and Virginia 1997), mites (Houck 1994), Collembola (Greenslade 1991) and bacteria (Henegan and Bolger 1996). Many works have included macroscopic non-insect groups within broader studies of soil fauna, although because of the difficulties with their identification (e.g. Duncan 1994), taxonomic resolution has often been simply to guilds (i.e. groups sharing a similar trophic niche) or families.

A core methodology that allowed comparison of studies across time and space might thus be provided by vegetation (for describing habitat characteristics) and invertebrate assemblages (for describing the greatest part of biodiversity). Where particular information is sought on other groups, these may then be related within a system context drawn from a comparable core methodology.

Stork and Samways et al. (1995) noted that ‘the indicator species approach has been criticised for its questionable assumptions, methodological deficiencies and sometimes biased application (e.g. Landres et al. 1988, Noss 1990), but that nevertheless it still has great value for monitoring certain changes that take place relative to the specific biodiversity question being asked’. Landres et al.’s (1988) critique was directed at use of vertebrate species as indicators, and so some of their comments may not be relevant in this section. However, the belief that one or a few species may be used to indicate everything, in all regions and systems, including relative biodiversity and effects of anthropogenic impacts, is obviously misguided. Species abundance is indicative of the extent to which the local environment meets the needs of a species. Lack of a single species is not indicative of anything unless this is viewed from a broader perspective which enables an interpretation of why the species is missing. Even then, the interpretation may only be indicative of environmental parameters pertaining to that particular species. In contrast, the use of species assemblages, when measured in a standardised manner, enable the comparison of a complete range of parameters associated with all the components. In addition, species responding similarly to particular environmental parameters may provide confirmation in the interpretation of the reasons for species abundance changes.

We can see that it is appropriate and necessary to include the invertebrates in measures of biodiversity, but problems revolve around the fact that we cannot hope to document the full immensity of their diversity. What is required is the use of a subset that may be defended as being representative of the high species richness and activity within this group of organisms, while still being practical in application. To be useful to managers, the subset would also have to be characteristic of easily recognisable vegetation types and dynamic stages.

5.2 USE OF INVERTEBRATES FOR INDICATING AND COMPARING BIODIVERSITY

New Zealand has high endemism in most invertebrate groups. Some higher taxons are particularly rich in this country e.g. Mollusca (Solem et al. 1981, Solem 1984), Collembola, Symphyla (Myriapoda), Chelonethi (false scorpions) and Opiliones (harvestmen) (Watt 1983). Other, larger groups, display an overall richness closer to the mean of New Zealand's proportion of the world's macro-invertebrate biota, but show particular richness and unique distributions in sub-groupings, e.g. Lepidoptera (Dugdale 1988) and Coleoptera (Watt 1983, Klimaszewski and Watt 1997).

The molluscs have received considerable attention in New Zealand, as they are amenable to litter sampling, and are generally identifiable by their shells. However, in a trial involving relocation of *Placostylus ambagiosus paraspiritus* on Cape Maria van Diemen in Te Pahi Farm Park in the far north, progeny of translocated snails were found to have different shell morphology from their parents (G. Sherley, pers. comm., August 1997). Molluscs are relatively sedentary, and therefore might be expected to reflect old associations. However this may result in slow responses to the rapid habitat changes.

In the far north, individuals of *Placostylus ambagiosus* have been found persisting under old remnant *Geniostoma ligustifolia* where these plants were apparently the only survivors of the former vegetation cover (G. Sherley, pers. comm., August 1997). Kuschel (1990) recorded similar persistence of cryptic, endemic fauna (beetles) in small indigenous vegetation patches in an Auckland suburb, suggesting that retention of even small fragmented indigenous habitat is a worthwhile conservation exercise.

Recent study by Barker and Mayhill (in press) in the East Cape region showed mollusc assemblages were related to particular indigenous vegetation assemblages, indicating a degree of habitat specialisation. However, they also found localised endemism to be very low, and that assemblages appeared to reflect a continual colonisation since the miocene. In contrast, limited Malaise-trapped beetle sampling from the same region provided c. 10% of species which were not only undescribed, but which were not represented in the National Arthropod Collection. This suggests that this group exhibits considerable local endemism. This is in accordance with conclusions from Kuschel's (1990) study in the Auckland region.

While it may be theoretically feasible to measure all organisms within a large system, it is impossible in any practical sense. This was well demonstrated by Disney (1986a) who calculated that given certain assumptions it would take about 2.5 millenia (at best) to map the occurrence of Diptera (flies) of Britain. One underlying assumption of the calculation was that of 1000 people being able to identify all Diptera to species, and all working continuously to achieve over 4 million identifications a year. He based his calculations on a 'colossal effort' that was put into a particular upland site in Britain, but he recognised that a meaningful interpretation of the significance of the findings required a similar effort on a range of other sites. He concluded that using inventory surveys as a tool for the identification of sites of conservation value was a lost cause.

New Zealand has a unique fauna, far less taxonomic expertise and a far shorter history of entomological inquiry than Britain. Crosby (1995) also concluded that comprehensive biodiversity surveys would be of limited value for comparing and monitoring changes over time within and between different areas in New Zealand.

5.2.1 Insects: the heart of biological diversity

Insects carry out a wide range of roles in terrestrial ecosystems. In addition to their consumption of live plant material, they are involved in pollination; leaf, litter and log breakdown; soil formation; general scavenging; parasitism and predation; as well as providing the dominant food for birds, lizards, and most freshwater fish (Watt 1975). Their overall effect, as is the case for mycorrhizal fungi (Miller and Jastrow 1994), is the retention of accumulated organic materials within biotic systems, through their control of the cycling of organic matter. They are a (and sometimes the) major influence on both the continuity and the adaptability of vegetation systems.

The total domination of terrestrial biodiversity by the insects demands their use if we are to characterise biodiversity. While other groups may be used to investigate particular interactions, until a relationship has been demonstrated with insect communities, they cannot be said to be indicators of biodiversity. In addition, we do not simply wish to indicate biodiversity, but to use the organisms as indicators of processes occurring in biological systems. The rich diversity of ecological roles provided by insects is thus indicative of the interpretative knowledge of systems that they may provide.

Of an estimated 20-30,000 species of insects in New Zealand (Watt 1982, Kuschel 1990, P. Johns pers. comm. 1994) only about half have been named and described to date (Watt 1982, Emberson 1994). However, for the same reasons that we need to include the invertebrates in biodiversity evaluations (i.e. the huge proportion of biodiversity that they comprise), we need to focus on the insects if we wish to evaluate biological diversity. We are therefore left with a need to devise a pragmatic approach to the problem of characterising insect communities.

The difficulties of measuring both the insect and fungal components of communities led Pielou (1993) to suggest that perhaps plants should be used as surrogate indicators of insect and fungal communities. However, Disney (1986a) noted there was limited hard evidence (e.g. Brooker 1982), and much anecdotal

evidence to suggest that some sites found to be poor in terms of floral diversity, score unexpectedly high in terms of the invertebrate fauna. In New Zealand, our ancient beech forests provide an example of this type of system. Although flora are species poor, the insect component of these communities appears at least as diverse as those of podocarp-broadleaf forests of the central North Island (Hutcheson unpublished data). Use of plants alone, before knowledge of these relationships is gained, would therefore be misleading.

Workers have used many different taxa and sampling systems in attempts to characterise terrestrial insect communities, or to investigate effects of habitat management on particular groups. Taxa used have varied widely, usually reflecting the researcher's background taxonomic experience (see New 1987) and experience with a particular method, or their observations of groups affected by particular management. Groups which are particularly favoured outside New Zealand include: ants (e.g. Anderson 1983, 1991, Anderson and Yen 1992, Altieri and Schmidt 1984, Majer 1978, 1983) and carabid ground beetles (e.g. Craig 1993, Niemelä et al. 1993, Vermuelen 1993, Lys et al. 1994).

Rather than simply provide a vast list of references to approaches made, at this point it makes better sense to attempt to first refine our requirements and evaluate the possible approaches further. We know that we need data that are characteristic of many complex systems to build predictive models, and that these are necessary for management of biodiversity at the landscape scale (Gordon and Hamilton 1994, Lautenschlager 1995, Watson et al. 1995). Therefore the approach used must be cumulative, so that studies are not isolated as they have been in the past. We also know that the complexity of the systems demands a pragmatic approach for the information to be usefully applied (Erhlich 1992).

The term 'pragmatic' is generally related to the financial resources made available, and these are currently extremely limited for actual studies on invertebrate communities. For a method of community characterisation to become accepted as a standard—which it must do if widespread comparisons are to be made—a few people must be able to compare two systems/treatments within a few months. Unfortunately, if we attempt to reduce the work input too much, we return to the problems of neither gaining much more knowledge of biodiversity, nor gaining sufficient understanding of system relationships to manage the resources sustainably. Let us first then consider some options for investigating insect communities, while constraining taxonomic, spatial and temporal protocols for the methodology.

5.3 POTENTIAL INDICATOR TAXA

Any potential indicator group must include a complete trophic range, so that the functional structure of the sample can better reflect the status of the insect community. While use of different functional groups for different studies has been suggested (e.g. White 1994), this would inhibit wider comparison between studies, and limit the ability to accumulate a knowledge base of system relationships and interpretive tools. Use of indicator species from a range of groups clearly requires a wider range of selection criteria, taxonomic skills and

interpretive tools. The use of a single multi-trophic group as a core basis for comparisons, appears to offer a more pragmatic approach.

To get to the heart of biological diversity, we need to consider species-rich groups.

Species-rich insect orders which comprise a full trophic range, include the Coleoptera (beetles), the Hymenoptera (ants, bees and wasps), and the Diptera (flies). These orders have been estimated to account for 23.8%, 7.7% and 7.1% respectively, of all presently described species of organisms on earth. Total potential numbers of species for these groups are estimated at 24.9%, 19.3% and 12.9% of a global total of some 12.5 million species (Hammond 1992). Although these figures are only estimates, they do indicate that the Coleoptera are better understood worldwide than either of the other two groups.

The Hymenoptera are not particularly well represented in New Zealand, probably because of our long isolation (Watt 1975), and they remain poorly documented (J. Berry, National Arthropod Collection, pers. comm. 1994). The Formicidae (ants) are a very obvious group of hymenopterans in arid and tropical zones (May 1989), where they are often used in community studies (e.g. Majer 1978, 1983, Anderson 1983, 1991). Their individuals (if such a term may be used for identical genotypes) form a large component of intensive surveys (Majer 1985a, 1987a, 1987b, Majer and Recher 1988, Majer et al. 1990, Recher et al. 1996). However ants are very poorly represented in New Zealand rainforest systems, with only nine endemics from a total of 38 species (Green 1993). This provides an excellent example of how methodology developed for systems different from those in New Zealand may be highly inappropriate for use here.

Many Diptera occur in large swarms that provide numerous individuals in samples, increasing the curatorial and taxonomic workload beyond the capacity of the worker resource available. Moeed and Meads (1987) recorded Diptera as accounting for 84.2% of individuals from total Malaise trap catches in the Orongoronga Valley. In contrast Coleoptera accounted for only 2.3%. These relative abundances resulted in the Coleoptera being identified to recognisable taxonomic unit (RTU), whereas the Diptera were poorly defined taxonomically.

The recognisable taxonomic unit is a tool utilised by entomologists when dealing with communities which contain many undescribed species. An example of this may be weevils (Curculionidae). We may be able to identify a weevil to subfamily or tribe, and very often to genus. Within a genus we may find we have many individuals which are undescribed, but taxonomically different from each other, i.e. putatively different species. These are separated and placed in a reference collection. In entomology the term recognisable taxonomic unit (RTU) is used to denote that this system is being utilised. Standardisation of these units across different studies is required in order to provide a 'common language' in the same way as Latin binomials do, to enable studies to be cumulative.

Didham (1992) sampled some individual tree canopies in the Kaikoura region with window traps, and found that Diptera accounted for 91% of total individuals, compared with only 1% for Coleoptera. Both orders were identified to RTU, but the Coleoptera were better able to be classified into meaningful

groups associated with sampling site. This was attributed to the Diptera being generally much more mobile than the Coleoptera.

The 1987 Australian Governmental Working Group on Biodiversity recommended the documentation of ants, Collembola and Coleoptera as indicator groups from the invertebrate fauna (J.D. Majer, Curtin University WA, pers. comm. 1991). Collembolla are very abundant in the litter layer, and include very few predacious species (Greenslade 1991), although they share a fairly ubiquitous distribution with the Coleoptera.

It has been estimated that beetles account for about 50% of New Zealand insect species (Watt 1982) which is similar to estimates for Australian insects of 40% (Lawrence and Britton 1991). Coleoptera range across all trophic groups and have been reported by several authors as being 'representative' of the species richness of insect communities (e.g. Moeed & Meads 1985, Neumann 1979, Tanaka and Tanaka 1982). The appropriateness and utility of the Coleoptera for representing insect communities has been recognised in some studies; detailed processing from a much broader sampling programme has subsequently focused on the beetle component, because of their high species richness, only moderate abundance, broad functional range, and better understood taxonomy and ecology (e.g. Hammond 1990, Moeed and Meads 1987, Neumann 1978, 1979).

5.4 SAMPLING

5.4.1 Sample collection

The numerous comparisons currently required to guide habitat management demand a relatively inexpensive standardised sampling system. The system must be: passive rather than attractive, so that samples may be validly defended as being associated with the habitat; continuous, in order not to be affected by episodic inclement weather; and independent of researcher and habitat type, so they may be applied consistently in any system. These criteria dictate the use of some form of passive interception trap, rather than e.g. light trapping (Holloway 1984), sweep netting (Wong 1984), quadrat counting (Topping and Sunderland 1992), branch clipping (Schowalter 1995) or counts of flying individuals (Pollard 1977). Because habitats change over time, historical species lists (e.g. Rushton 1987) do not provide the direct link with current habitat that is necessary to guide habitat management. Accurate and standardised habitat records are vital to form the links between habitat and biota that are required. Museum specimens have also been suggested as an approach; however, these generally do not contain records of the vegetation type at the time of capture. In addition, rare species are often over-represented, and common species under-represented in such collections. The major role for museum specimens is to form the basis of accurate taxonomy (section 5.4.3).

Traps need to be cheap, non-powered, easily transported and installed, and able to be left unattended. A number of traps fit this description, including pitfall, window, Malaise, and various combinations of these. Disney (1986b) suggested using a set of various trap types in order to sample both ground-moving and

flying insects, an approach supported by Basset (1988) and New (1987) who also commented that every insect ecologist would favour a familiar taxonomic group. Diamond and Case (1986) acknowledge that multiple methods are appropriate to community studies, but they make a plea for a modest number of approaches, so that broad standardisation (and therefore comparison) is made easier. Ehrlich (1992) recognised the necessity of pragmatic approaches to biodiversity studies because of the urgent requirements of conservation management, and recommended restricting samples taxonomically and spatially. As the objective for habitat management-related studies is to obtain characteristic samples from the community rather than to trap as many organisms as possible, a single trap system is favoured over composite traps.

Historically, pitfall traps have been used very extensively for sampling ground-moving arthropods, particularly in the Northern Hemisphere. Consequently, a large information base, in the form of collections and taxonomic and ecological knowledge, is now available to workers in that area of the world. The positive attributes of the Coleoptera have been recognised and many pitfall studies have focused on the Carabidae (ground beetles) which form a prominent part of the catch (e.g. Refseth 1980, Day and Carthy 1988, Luff et al. 1988, 1992, Desender and Turin 1989, Eyre et al. 1990, Niemelä 1990, Rushton et al. 1990, Gardner 1991, Fuisz and Moskat 1992, Niemelä et al. 1993, Walsh et al. 1993). Pitfall traps are simple to replicate, and Carabidae taxonomy and ecology are very well understood in the Northern Hemisphere. These two factors together with their long historical use continue to influence workers in insect community ecology throughout the world. However pitfall catches are usually quite limited in numbers of both individuals and species, and carabids are generally only predatory, so the trophic structure of a community may not be well reflected in samples. In addition, the Carabidae are not as well represented or known in New Zealand as in the Northern Hemisphere, and endemic species are generally forest-dependent (A. Laroche pers. comm. 1994). The latter reflects the original vegetation cover of New Zealand, and may restrict the ability to use carabids in a standardised approach across non-forest habitats. Other groups, particularly the amphipods, dominate the litter fauna in this country.

For over 30 years entomologists have expressed reservations about the use of pitfall traps (Greenslade 1964, Southwood 1978b). In many pitfall studies, all taxa caught are included in analyses in an attempt to utilise the largest possible subsamples. This adds difficulties in New Zealand where many of the cryptic ground fauna are undescribed or unknown. In addition, Collembola, Amphipoda and mites numerically dominate the catch. The Collembola are known to respond to CO₂, which is given off by disturbed organic soil, thus confusing interpretation of pitfall catches. Because of their variable size, morphology and markings, the Amphipoda are a particularly difficult group to rapidly identify (Duncan 1994). This is true also of the mites, which are poorly understood in New Zealand (J. Clarke pers. comm. 1994). These constraints limit the accuracy and interpretability of these groups for pragmatic invertebrate community characterisation.

Anderson (1991) found that the relative abundance of savannah ant species captured in pitfall traps was highly correlated with quadrat counts, but Topping and Sunderland (1992) compared pitfall and absolute density sampling of spiders and found differences in numerical species composition, sex ratios and

fluctuations of single species over time. They concluded from their study that there were severe limitations to the application of pitfall traps. Yen (1987) found the correlation between classifications of pitfall-trapped Coleoptera and 8 vegetation communities stronger than that found with vertebrates, however the beetle classification did not discriminate well between perceived habitat types. Pitfall traps are not recommended for comparing samples obtained at different times, and Sanderson et al. (1995) reasoned that pitfall traps are unreliable for obtaining quantified information. This limits data from them to presence/absence, which are inadequate for elucidating changes in species dominance in response to habitat change or management regime.

Hosking (1979) compared light, sticky, window and Malaise traps. He found that the Malaise trap (Townes 1972), the catch of which is dominated by low-flying insects, to be most efficient for capturing flying Coleoptera. Malaise traps caught almost twice the number of beetle families than the other traps. Window traps delivered a disproportionate number of individuals from single species of Colydiidae and Scolytinae, which tend to 'drop' on interception. By contrast, individuals captured in Malaise traps move upward toward the light. Sticky traps are not suitable for sampling where there are difficulties with identification, or interest in the curation of specimens. Hansen (1988) found Malaise traps to be the most convenient of a range of trapping methods. Malaise traps operate passively and continuously, are non-powered, cheap, easily transported, erected and serviced, and weekly catches are large enough to be robustly classified with cluster analysis (Hutcheson 1990, Hutcheson and Kimberley in press).

The correlation between the abundance of Coleoptera and that of other invertebrates in catches varies with different trapping methods (Table A, overleaf). Full-size Malaise traps show a high correlation between beetle abundance and total arthropod catch. Apart from Moeed and Meads' (1984) cone intercept trap correlation, higher sampling effort provides higher correlations, as illustrated by pitfall trapping (Table B, p. 47). Comparison between full-size and half-size Malaise traps (see Table A) also demonstrate a similar rise in this correlation with increased sampling effort. Hutcheson (1996, unpublished data) obtained much higher catches from Malaise traps erected on the ground than those at 12 m within forest. Moeed and Meads' (1984) cone intercept traps were set at 3, 12 and 24 m above the ground, and they found little difference between activity at 12 and 24 m. This suggests that the main decrease in insect activity may be between 1 and 3 m from ground level. Although catch sizes from Moeed and Meads' cone traps were very limited, these authors did suggest that beetle activity may be more highly correlated with total activity where the soil fauna representatives are not part of the catch.

The following tables (overleaf) were originally given in Hutcheson (1996: Table 5a and 5b) as beetles versus total catch. Removing beetles from the total catch column reduced most correlations slightly and increased the relative ranking of Malaise traps.

The effects of insufficiently sized samples were demonstrated in a study where paired sets of full size and c. 5/8 size Malaise traps were erected in four habitat types (Dugdale and Hutcheson 1997). Beetle catches from the smaller traps could not be discriminated according to habitat type whereas those from full size traps were. Traps orientated with the collecting jar southward (i.e. away

TABLE A, ADAPTED FROM HUTCHESON 1996: CORRELATION OF BEETLE ABUNDANCE TO TOTAL ABUNDANCE OF THE REST OF THE CATCH IN A RANGE OF STUDIES, IN DESCENDING ORDER OF R.

REFERENCE	SAMPLING METHOD	VARIATION	COUNTRY	R
Moeed & Meads 1984	Cone intercept	Height	NZ	0.99
Moeed & Meads 1987	Full size Malaise	Temporal	NZ	0.89
Neumann 1978	Full size Malaise	Spatial	east Austr.	0.86
Norton & Walsh 1994	Pitfall	Spatial	NZ	0.84
Hammond 1990	Sheet intercept	Spat/temp	North Sulawesi	0.83
Wong 1984	Sweep net	Spat/temp	west Malaysia	0.80
Tanaka & Tanaka 1982	Sweep net	Temporal	West Indies	0.77
Moeed & Meads 1987	Emergence	Temporal	NZ	0.69
Majer & Recher 1988	Pyrethrum	Spatial	W A & Vic. Austr.	0.68
Levings & Windsor 1982	Litter	Temporal	Costa Rica	0.68
Wong 1984	Sweep net	Spat/temp	west Malaysia	0.56
Hammond 1990	half-size Malaise	Spat/temp	North Sulawesi	0.46
Moeed & Meads 1986	Litter	Spat/temp	NZ	0.45
Moeed & Meads 1983	Trunk traps	Spatial	NZ	0.43
Hammond 1990	Baited pitfall	Spat/temp	North Sulawesi	0.30
Majer et al. 1990	Pyrethrum	Spatial	W A & Vic. Austr.	0.25
Majer & Recher 1988	Pyrethrum	Spatial	W A & Vic. Austr.	0.25
Moeed & Meads 1985	Pitfall	Spat/temp	NZ	0.24
Abbott et al. 1992	Branch clipping	Temporal	West Austr.	0.23
Majer & Recher 1988	Pyrethrum	Spatial	W A & Vic. Austr.	0.10
Neumann 1991	Pitfall	Spatial	east Austr.	0.02
Majer et al. 1990	Branch clipping	Spatial	W A & Vic. Austr.	-0.66
Hammond 1990	Unbaited pitfall	Spat/temp	North Sulawesi	-0.89

TABLE B, FROM HUTCHESON 1996: CORRELATION OF BEETLE ABUNDANCE TO TOTAL ABUNDANCE OF THE REST OF THE CATCH, FROM UNBAITED PITFALL TRAP STUDIES.

AUTHOR	R	TRAP SIZE (DIAM.)	NO. TRAPS PER SITE	NO. SITES	PERIOD LENGTH	NO. PERIODS	NO. SAMPLES
Norton & Walsh 1994	0.84	65 mm	100	4	4 weeks	13	5200
Moeed & Meads 1985	0.24	75 mm	10	4	1 month	13	520
Neumann 1991	0.02	18 mm	20	4	2 weeks	36	2800
Hammond 1990	-0.89	75 mm	20	3	1 week	12	720

from the strongest light source) within forest cover have also been found to provide low catches (Hutcheson 1990). Presumably, this was because the phototropism of intercepted insects was not fully utilised in their capture. These low catches were also more difficult to classify correctly according to site. These studies provide indications that a reasonably high sampling effort is required for beetle catches to be characteristic of the wider community, and that full size Malaise traps, used appropriately, are able to accomplish this.

Flight interception traps separated from the ground do not sample ground movers and emerging insects, whereas such species are represented in Malaise catches in addition to insects flying within about a metre off the ground. New Zealand has a large endemic component of flightless beetles and these should be included in samples as they can strengthen the local component, and therefore site specificity, of catches. Malaise traps, which sample within about 1 m of the ground, are thought to capture about 80% of the very mobile members of the family Staphylinidae (J. Klimaszewski pers. comm. 1994).

The area from which Malaise traps capture beetles depends on the mobility of the beetles, which varies with species and life history stages, and trap placement. The trap depends upon individuals being positively phototropic, which is characteristic of beetles during the dispersal phase after emergence. Hutcheson and Kimberley (in press) found they were able to discriminate between Malaise samples from traps in different habitat types approximately 50 m apart, whereas Dugdale and Hutcheson (1997) found substantial influence on beetle assemblage composition from a different habitat type within 20 m. This suggests that the majority of the catch in Malaise traps may be sourced from within c. 3/4 ha. This approximates the spatial scale at which Økland et al. (1996) were able to relate saproxylic beetles to habitat characteristics using window traps.

Current information thus indicates that full size Malaise traps sample a broad range of Coleoptera species with a wide range of mobility; that classification groupings are compatible with habitat patch dynamics and management; and that these catches correlate well with the broader activity of insect communities over both time and space.

5.4.2 Timing of sampling

Underwood (1993) reasoned that timing and frequency of sampling should be determined by logical thought about the processes operating and the organisms being examined. For example it would be unproductive to attempt to characterise New Zealand insect communities in winter when they are most active in summer. In New Zealand, the activity patterns of adult beetles fluctuate over the spring to autumn seasons according to each species biology (Moeed and Meads 1984, 1985, 1986, 1987; Hutcheson 1990), however expense prohibits all seasonal permutations being represented in a pragmatic sampling programme. If the communities are sampled randomly in time, we may collect data which are a greater reflection of time of year than association with habitat (Hutcheson 1990). Annual adult insect activity in New Zealand peaks between spring and autumn, and Hutcheson (1990) suggested that standardisation on a short period of sampling within this time may be a cost-effective approach.

When a full spring to autumn season of weekly Malaise trap catches of Coleoptera were subjected to divisive polythetic classification (TWINSPAN), catches were grouped by site, time of season and micro-site in descending order of influence (Hutcheson 1990). Subsequent work (Hutcheson and Kimberley in press) showed that catches from various habitats could best be discriminated when sampling was restricted to a four-week period within December in comparison with sampling from longer periods. The improved homogeneity of catches from each trap during this period is due to lower catches from the beginning and end of the activity season not being included, and also a reduction in the influences of seasonal population changes. Because the four-week sampling window falls during the beginning of the beetle activity peak, the confounding influence of increasing ranges of very mobile species, which occurs later in the season, is also reduced.

5.4.3 Level of taxonomy required

Ehrlich (1992) emphasised that pragmatic approaches are required for comparative studies of insect communities because of community complexity and also because of the pressing nature of the problems of ecosystem management, particularly those concerned with biodiversity conservation. The more modest the sampling and taxonomic input required, provided it is soundly based, the more likely it is that understanding drawn from a broad spectrum of habitat types and dynamic stages may be applied to management situations.

Warwick (1993) discussed taxonomy to family level in marine studies and reported little information being lost relative to the time saved and lower level of expertise required. However, for the data to be useful, the level of taxonomy must enable functional groups to be discriminated from within the community. Taxonomy to this level is necessary because mechanisms of population limitation may be fundamentally different for different functional groups (MacFadyen 1980, Van Emden 1990, Wiegert and Owen 1971). Taxonomy to at least generic level is necessary for discrimination between trophic levels in the Curculionidae (weevils), the dominant family of Coleoptera. Use of standardised RTUs (i.e. to named species as far as possible), allows access to published and unpublished ecological information. A major advantage of this approach is that as additional taxonomic and ecological knowledge becomes available, it may be incorporated into data revisions and subsequent analyses.

Taxonomic determination only to guild level provides the possibility that dramatic change in the abundance of species may go undetected as a result of counterbalancing between abundance increases and decreases of different species within guilds. Hutcheson and Jones (in press) in a study of beetles within radiata pine stands, found such 'within-guild' replacement at different stand ages, indicating that for this reason also, taxonomy simply to guild is inadequate.

Oliver and Beatie (1993, 1996) tested the relative accuracy of taxonomy by specialists and 'parataxonomists' (those with less formal training) for rapid biodiversity assessment. They found that different groups of organisms were identified with varying accuracy, and that there was a greater tendency for less experienced workers to elevate variation within species to RTU status. Different groups of Coleoptera vary widely in the ease with which they may be identified. It should be noted that within genera where species radiation is occurring, even species defined by specialist taxonomists may be arbitrary, or even temporary, divisions in a gradient of genetic change (O'Hara 1994).

Hutcheson (1990) subjected subsets from spring to autumn sampling to classification analysis. This showed that in order to reduce the sampling period to a minimum and still retain catch discrimination, taxonomy had to be at least to RTUs. Taxonomy to species level delivers the most ecologically useful information, as the functional structure of the community may be defined by component species ecology where these are known. Dissection of genitalia, the standard approach to separating difficult species complexes, is presently impossible for insect community studies, where there may be large numbers of specimens and a very limited number of experienced workers. The compromise solution is lumping such difficult groups into single RTUs, which are identifiable from their exterior morphology. These may then be standardised at a national level to allow comparisons between different studies. Such groups may receive special taxonomic attention at a future time.

5.5 MALAISE TRAPPED BEETLE SAMPLES AS BIODIVERSITY INDICATORS

Stork and Samways (1995) provided a list of indicator species requirements against which Malaise trapped beetle catches are evaluated below:

- high taxonomic and ecological diversity (many species in each locale or system);
- close association with and identification with the conditions and responses of other species;
- high ecological fidelity;
- relatively high abundance and damped fluctuations (i.e. they are always present and are easy to locate in the field);
- narrow endemism, or if widespread, well differentiated, either locally or regionally;
- (relatively) well known taxonomy and easy identification;

- (relatively) good background information (e.g. on genetics, behaviour, biochemistry, ecology, biogeography);
- large random samples encompassing (large) species variation are possible;
- functional importance within the ecosystem is (relatively well) understood;
- (relatively) predictable, rapid, sensitive, analysable, and (?) linear response to disturbance. Some populations are known to respond to environmental change.

We can see that Malaise-trapped beetle catches meet these basic criteria well. Even where occasional attributes are not a perfect fit for some species, the use of assemblages allows an excellent fit, particularly relative to most other groups and approaches.

5.6 ANALYSES OF INSECT COMMUNITY DATA

A primary requirement for applied insect community studies is that the researcher should be able to discriminate between samples from different vegetation systems. An inability to discriminate samples from obviously different vegetation systems could be due to: the insect communities not being different, the sampling being insufficient to represent the communities, or to the processes controlling the insect communities operating independently of the vegetation. The world-view of ecologists and resource managers is based upon obvious physical and vegetation characteristics. It is thus necessary to link vegetation to insect samples for these to be usefully included in resource management.

At the level of current understanding, insect community sampling is initially a descriptive rather than a mensurative exercise. The spatial heterogeneity of natural insect communities is not randomly distributed, but controlled by a complex of both external and internal, mostly non-random interacting influences (Perry 1989). This does not easily enable sampling of these communities to be subjected to control, replication, randomisation and interspersions; therefore initially, use of inferential statistics based on normality is inappropriate (Hurlbert 1984, Hurlbert and White 1993). However this does not mean that experimental sampling should not proceed (Hurlbert 1984). The investigation of natural phenomena must begin with a descriptive phase followed by the formulation of testable hypotheses which can then be subjected to a more rigorous mensurative experimental approach, tracking the 'what, how, and why' sequence of investigation (Wiegert 1988). Wiegert (1988) observed that 'the null hypothesis, which assumes that there is no explanation for observed phenomena because they are entirely random, and which is such a powerful tool when interpreting the results of mensurative experiments, does not assist in the formulation of ecological hypotheses'. Hurlbert (1984) drew the attention of the ecological community to the inappropriate use of inferential statistics, and commented that lack of statistical tests for significance was preferable to their misuse. The practice is still continued however, as pointed out by Hurlbert and White (1993).

5.6.1 Diversity measures

Diversity may be assessed at various scales along gradients of genetic information, spatial distribution and time, and the resulting bewildering multitude of apparently valid approaches has contributed to a lack of a standardised method for comparing biodiversity (Gordon and Hamilton 1994, Hill 1973). Diversity has often been used interchangeably with species richness (i.e. number of species), and this measure is accepted by some workers as the simplest diversity index. However it is the measurement of both the number of species, and how individuals are distributed among these (i.e. equitability) that has generated the plethora of diversity indices.

Although in most communities all species cannot be sampled (Lautenschlager 1995), Palmer (1990) reported a correlation between species observed (SO) and true species richness (SR) of 0.973, implying that SO may be used to estimate SR providing that the sample size remains constant (Mathers 1992). However, sample size was interpreted in terms of total numbers of individuals, suggesting there are difficulties in applying this relationship to many real sampling situations.

Species in communities occur in a range of abundances which may be characteristic of different communities (Pielou 1975). In an effort to better characterise the structure of communities for comparative purposes, indices of diversity attempt to incorporate both species richness and their population abundances. In this form they can be thought of as measures of 'qualitative dispersion' in the same way as range, standard deviation and variance can be thought of as measures of quantitative dispersion (Kotz and Johnson 1982). It should be noted that such diversity indices treat all species as functionally equal. It has already been pointed out in sections 3.4 and 3.5 that resource managers need qualitative information (e.g. endemism) in addition to quantitative information, and that this is true also for ecological studies.

Simpson's index (D), or its reciprocal, measures the likelihood that two individuals selected at random will belong to the same species (Peet 1974). It has a limited power of discrimination where there are large numbers of rare species, being most strongly influenced by the more abundant species. The Shannon-Wiener or (Shannon's) index has been derived for both infinite (H') and finite (H_B = Brillouin index) communities (Pielou 1977). It was derived from information theory and gives the likelihood that an individual selected at random will belong to a particular species. It is influenced mostly by species of intermediate abundance. These two indices plus species richness (S) were used by Hill (1973) as the diversity numbers N_0 (S), N_1 (eH') and N_2 ($1/D$), to give indications of the influences of rare, intermediate and common species in a community. Magnussen and Boyle (1995) recently concluded that for the same statistical resolution, the Simpson index requires about nine times as many samples as the Shannon-Wiener index. The Berger-Parker index (d) has also been favoured by some authors (e.g. Gordon and Hamilton 1994), but has been reported as being less effective in discriminating differences in entomological datasets than H' (Tonhaska 1994).

Just as quantitative measures of dispersal (or of central tendency) can give inconsistent orderings, so too can different diversity indices (Kotz and Johnson 1982). Because diversity indices combine species richness and their

abundances, they are often combined with a measure of equitability (the distribution of the individuals in the sample among the species present). For a given species number, evenness (J) reaches its maximum when all species are present in equal numbers (Pielou 1975).

Problems in obtaining meaningful information about communities from these diversity indices are themselves diverse. They range from the fact that populations are not randomly distributed either spatially or temporally, through the lack of spatio-temporal definition of communities, with each component species constrained according to a range of factors. However the main difficulty with diversity indices stems from the fact that a single value of a diversity index contains less information than does the full species abundance pattern from which it was derived. Composite diversity indices confound species richness, species abundance, component species and species function, and have been found to be of limited use in several studies (Hutcheson 1990, Samways 1984, Tonhaska 1994).

Species abundance contributes to community structure and may reflect some aspects of communities at the systems levels that are used by land managers. However, communities involve vast numbers of diverse organisms whose organisational pattern is controlled by the interaction of component structure and function rather than simply by their relative abundance. This is discussed by Wiegert (1988), who suggests systems may be classified into groups by their 'structural characteristics' (within which he included the community components), and whose functional characteristics are determined by all the components. Tokeshi (1993) notes that the imbalance of attention given to diversity indices is due more to conceptual appeal than to any scientific rigour or superiority.

The concept of diversity indices still holds powerful sway in ecological studies, and a simple, single figure diversity comparison is often sought by managers who are unaware of the shortcomings of the approach. A distinction between a cardinal approach to diversity such as delivered by the diversity indices mentioned above, and an ordinal one where individual attributes of component species are included, was recognised by Cousins (1991) who felt that although cardinal indices are widely used for the environmental assessment of areas of habitat, ordinal indices offer many advantages. Ordinal indices can include recognition of species, functional groups or classes of species abundance.

A simple measure of diversity was derived by Hutcheson and Kimberley (in press) using a summation of the abundance classes (SAC) which were found to deliver optimum sample discrimination in TWINSPAN analysis by Hutcheson (1990). These classes form the default option of the classification procedure, which was designed for vegetation analysis, suggesting that they are widely applicable to ecological data. As such they may be more appropriate for abundance data transformation than the more usual log transformation. The 5 classes defined are: class 1 = 1 specimen, class 2 = 2-4 specimens, 3 = 5-9 specimens, 4 = 10-19 specimens and 5 = 20+ specimens.

Summed abundance classes (SAC) give similar patterns of diversity for different samples when compared to established measures of diversity (Hutcheson 1996). SAC is dominated by species richness rather than high abundance of individuals within species, reflecting the general acceptance that species richness is the

area of diversity that is most important. This is also demonstrated by the emphasis shown in e.g. Shannons' index. Because abundances within SAC are associated with component species, it is relatively easy to compare diversity within subsets such as trophic groups or families of beetles. This then may be considered a more robust system for ecological datasets. Like all diversity indices, however, only limited understanding of the communities is possible from the index alone. Reciprocal changes in species richness and abundance, which may be pertinent to understanding of community response to environment and habitat management, may still remain undetected, as demonstrated by Hutcheson and Jones (in press).

5.6.2 Species abundance patterns

Patterns of species abundances and the development of models which appear to fit the patterns observed, have historically also preoccupied community studies (e.g. May 1975, Pielou 1975). These models reflect attempts to describe abundance relationships in a way that would assist understanding of how communities are organised. Motumura (1932) suggested the geometric series as an empirical relationship in the first model proposed. This was followed by the log series (Fisher et al. 1943), the log normal (Preston 1948) and various others including the 'broken stick' of MacArthur (1957), which have attempted to link abundance patterns to the niche concept.

The niche is a function of, and inseparable from, the single species concept, but is not a tangible finite resource. The community has a large influence on the concept of the niche. This varies with the flux of different component populations at a range of spatial and temporal scales dependent upon a large number of factors. For example, there are increased opportunities for epiphytic growth with increases in vegetation structure, and parasitic relationships between endophytic mycorrhizae and monocot plants tend toward mutualistic relationships over evolutionary time (Clay 1990). Several other models, including both neutral (niche independent) and composite (encompassing two or more assembly rules), have been proposed in attempts to encompass these concepts. Tokeshi (1993), in his review of this whole area, makes the point that if species abundances are dependent upon the relative abundances of different resources, which are in turn influenced by a range of loosely related factors, specifying a niche-division rule (i.e. model) may not be a particularly illuminating exercise.

Tokeshi (1993) showed that the geometric and the log series may be nested inside the log normal series, suggesting that the different models may be a function of definition of community boundaries (i.e. scale) rather than community assembly rules. While the log normal curve has been found to fit many datasets (Hughes 1986), if not most (Zak 1992), May (1975) gave it careful consideration and concluded that this pattern was essentially a statistical consequence of large numbers. This supports Tokeshi's (1993) suggestion that abundance distribution models are consequences of the definition of community boundaries. Portraying rank-abundance may depict dominance patterns of abundance for comparisons (Samways 1984), although in support of May (1975) these were found to be of little use for recognition of e.g. Malaise-trapped beetle samples from different communities (Hutcheson 1990). Consequently species abundance models are not suggested as being useful.

5.6.3 Multivariate techniques

The need for a means of objectively comparing communities using data that are bulky, complex and noisy, which show redundancy, internal relations and outliers, and for which some information is only indirectly interpretable, without discarding information—together with the increasing availability of computing facilities—has fostered the development of multivariate methods based on comparisons of sample/species matrices. The extensive literature on the collection and objective analysis of vegetative community data, where sites are either grouped (classification) or ordered (ordination) based upon internal attributes (e.g. Whittaker 1973, Mueller-Dombois and Ellenberg 1974, Gauch 1982, Greig-Smith 1983, Kershaw and Looney 1985), has provided a background for the wider use of these techniques in other areas of ecology such as insect community analysis. Multivariate methods can be divided into three groups: direct gradient analysis or regression analysis, indirect gradient analysis or ordination, and classification or cluster analysis.

Gradient analysis

Gradient analysis attempts to arrange samples along one or more environmental or compositional gradients (Whittaker 1973). Analysis is termed direct, where samples are arranged along an environmental gradient (Whittaker 1982), or indirect where species composition is used to define axes of change. With direct gradient analysis each species is treated individually as a response variable to one or more explanatory environmental variables. With indirect gradient analysis, axes need not directly correlate with perceived environmental gradients, however the technique is usually used in vegetation analysis to identify community composition change associated with environmental factors (Austin 1976). A wide range of techniques are available for indirect ordination. These include methods such as polar ordination (PO) (Bray and Curtis 1957), principal component analysis (PCA) (e.g. Greig-Smith 1983), reciprocal averaging or correspondence analysis (RA) (Hill 1973), detrended correspondence analysis (DCA) (Hill & Gauch 1980, Hill 1979a) where distortions of RA are detrended, and multidimensional scaling (MDS) (Kruskal 1964) which uses rank order information from a similarity matrix.

DCA is reported as usually being better than other multivariate methods in modelling difficult datasets, although outliers causing discontinuity are problematical, and are best left out of the analysis if good justification can be provided for this (Gauch 1982).

Like other multivariate analyses, DCA reduces the multidimensionality represented by multiple species to a limited number of gradients. Separation of samples within these reduced dimensions is related to the similarity of their original species composition, making them comprehensible to a human interpreter (Gauch 1982). DCA corrects two problems present in Reciprocal Averaging (RA)/Correspondence Analysis (CA). These are: a compression of the ends of the first axis, and an arch distortion of the second axis, caused by its orthogonal relationship to the first axis (Kent and Coker 1992). DCA scales small segments of the axes in units of the average standard deviation of species turnover, such that equal distance of ordination corresponds to equal difference in species composition. Eigenvalues associated with each axis indicate how well

the axes model the dominant cause of variation. Four main axes of variation are supplied by the analysis procedure, with their ranking relating to the degree of variation in the data accounted for by each axis. Plotting of the first two axes is usually sufficient to indicate patterns of sample groupings where these exist.

Classification (Cluster analysis)

Of the variety of classification techniques for community data, two methods of polythetic hierarchical clustering have been widely used. Agglomerative clustering has been applied to grouping both community data and taxonomic characters (Goodall 1973, Sneath and Sokal 1973). However, the approach that has shown greatest promise for ecologically meaningful analysis of large entomological datasets is divisive classification as used in the program TWINSpan (Hill 1979b). This approach uses ordination to arrange all samples along a gradient which is then progressively divided. Hill recognised the need for a program which would render datasets comprehensible and sensible to ecologists.

A major advantage of this approach when it was first developed was that it is extremely efficient of computer memory. This is because all information is included at the beginning of the process, the most important divisions occur first, and divisions may be stopped as soon as groupings become ecologically meaningless. Gauch and Whittaker (1981) found the divisive method 'usually more informative' with vegetation field data, and species assemblage identifications using TWINSpan are now apparent in several fields, including that of insect community analysis, although generally with water or ground beetle data (e.g. Rushton 1987, Eyre et al. 1992, Luff et al. 1992, Foster 1993).

Hutcheson (1990) found that rank abundance graphs and the diversity indices discussed by Hill (1973) (i.e. number of species, $\exp(H)$, and Simpsons index/1) were unable to discriminate between Malaise-trapped beetle samples from two different habitats. In contrast, TWINSpan showed the two communities to be clearly different at the first division.

Neither cluster analysis nor ordination uses the sampling structure (i.e. replicates within different sites, times etc.) but simply displays the sample unit distribution graphically, allowing visual evaluation of ecological meaningfulness. Clarke (1993) discussed statistical tests of significance of multivariate data investigating anthropogenic impacts on marine systems and noted that the classical multivariate equivalent to analysis of variance (MANOVA) is inappropriate for many datasets because of the high presence of zeros in the matrices. He argued for a simple ranking (i.e. distribution-free) approach and suggested a test based not on the distances between samples in an MDS plot, but on the corresponding ranked similarities in the underlying triangular similarity matrix when analysing for changes in community associated with anthropogenic influences.

Given the current lack of knowledge of the driving influences on insect biodiversity, the oversimplification and transformation of data should be viewed with some caution. The benefits of viewing raw and summarised data (e.g. from TWINSpan), in the context of habitat documentation, and where interpretations may be drawn from the component species ecologies, can return

much more understanding of why ecological processes occur in the manner observed, than with many other approaches.

5.6.4 Use of classification to refine sampling protocols

The primary objective for management related studies, is to obtain characteristic samples from perceived habitats as cheaply as possible. This requires determination of the parameters that best enable samples from various habitats to be discriminated with the minimum expenditure of effort. Insect communities change markedly during the spring to summer period, as the emergence and activity of various species rises and falls according to their biology. This seasonal variation imposes a problem for community comparison, as it may be at least as large as spatial (habitat) variation. The problem was addressed by Hutcheson and Kimberley (in press) who subjected weekly catches from three Malaise-trapped beetle sampling series to TWINSpan classification. Four main habitats were sampled, and samples included catches from similar habitats separated by 100 km, and by 4 years.

In the study, various timing subsets were tested from the total dataset, to determine when best discrimination to habitat type occurred, and how long this period should be.

Catches from a short period immediately prior to peak adult activity were generally better classified into ecologically meaningful groups, and returned higher Eigenvalues (amount of the variance accounted for by the division) than catches from a longer period. Because of occasional anomalous classifications, it was necessary to include a group of catches from each trap to corroborate the classification groupings. Four consecutive weekly catches/trap gave best discrimination. This was also the number of samples found to be necessary to adequately represent stream invertebrate communities for ordination and classification purposes in Melbourne, by Marchant (1990). In contrast to Marchant's findings, the terrestrial invertebrate communities showed decreasing Eigenvalues and clarity of classification as catch number increased above four per trap. This was due to the increase in catch heterogeneity incorporated into the database with the extension in time, reflecting the more complex and variable nature of the terrestrial insect community, habitat and environment.

Use of the sampling protocols defined by Hutcheson and Kimberley (in press) allows sampling to be reduced to a pragmatic level, while being optimised for sample discrimination. The high correlation found between beetles and total insect abundances from Malaise trapping (Table A) gives some confidence that the Malaise-trapped beetle samples from this period provide activity attributes that are characteristic of the insect communities from which they were captured.

In addition to reducing sampling to a pragmatic level, the four consecutive weekly catches also provide a relative measure of significance of difference (R. Littler, Univ. Waikato, pers. comm. 1995). The four weekly catches from each trap provide a measure of temporal variation, against which catches from other traps (i.e. the spatial variation) are automatically gauged in the classification procedure. This provides the measure of 'ecological meaningfulness' with which divisions in the TWINSpan classification are judged. Thus if all or most of

the four weekly catches from each trap are grouped together, it may be concluded that the influence of spatial variation outweighs that of temporal variation. If these 'trap sets' become dispersed in the divisions, the temporal variation is likely to be as important as that due to trap siting. This measure may be relative to the size and discontinuity of the data set however, being less able to discriminate clearly where variation in a large data set is subtle and continuous (Hutcheson and Kimberley in press).

5.6.5 Functional (trophic) structure

The aspect of the communities which is perhaps of most interest, and which has an integral association with the species present and their relative abundances, is the function of the species within the community. The explanatory potential of the full trophic range of the beetles is a major argument for their use in ecological studies, and also for using taxonomy that is as detailed as is pragmatically possible. The search for and explanation of pattern that forms the essence of community ecology, is dependent upon the parameters of species composition, the nature and form of relationships, and their dynamics in space and time (Putman 1994). Trophic structure must therefore be an integral part of the description of the communities sampled, allowing comparison of their similarities, and perhaps more importantly, their differences.

Although beetles are better known than most species-rich insect groups, knowledge of the larval biology of the majority of the Coleoptera is still sparse, and is usually predicted from taxonomic relationships. Trophic descriptors are both necessary, and at the same time constraining, as in order for descriptions to be understandable, descriptors must be simplifications of actual functions, even where these functions appear well known (Pianka 1980).

Phytophagous beetles have often been implicated in tree deaths, when beetle development is actually dependent upon tree health debilitation (e.g. Schowalter and Filip 1993). Where vegetation is not subjected to extreme violence (e.g. blowdown, snowbreak, fire, harvest), a long decline to tree death occurs, with a gradual loss of vigour and an associated reduced ability to counter disease and/or insect attack (Hosking and Hutcheson 1986, 1999, Hosking 1993). A range of beetle species interacts with this gradient at all points and in a variety of combinations with other organisms, extending from attack of apparently healthy tissue in association with endosymbionts such as protozoans, into invasion of areas of advanced cellulose breakdown in symbiotic associations with a range of fungi (White 1993). Drawing a line between primary consumers and detritivores is thus both difficult and necessarily artificial.

5.7 CONCLUSIONS AND RECOMMENDATIONS FOR NEW ZEALAND

Evaluation of the various types, extents and management of vegetation systems for their ability to retain endemic biodiversity requires a standardised measure of insect assemblages to be applied across a range of systems. Hypotheses raised from interpretation of this data then need to be tested in a geographically expanded programme.

Methodologies developed for systems different from those in New Zealand may be highly inappropriate for use in this country. Malaise-trapped beetles are proposed as the appropriate approach to use to evaluate invertebrate biodiversity for the following reasons:

- invertebrates provide the majority of biodiversity;
- insects provide the majority of invertebrate biodiversity;
- beetles provide the major group of insect biodiversity;
- beetles range across all trophic groups and habitats;
- Malaise-trapped beetles show high correlation with activity of other insect groups;
- Malaise-trapped beetle catches, in association with divisive classification, provide a workable and tested method which gives repeatable results and has been developed for the New Zealand situation;
- results to date reveal a close relationship between Malaise-trapped beetles and the attributes of their habitat.

Research into the conformity of conclusions drawn from beetles with those taken from other insect taxa is also required. This is assisted by the Malaise trap approach, which samples a broad range of insect groups. A preliminary study which included both Lepidoptera and Coleoptera (Dugdale and Hutcheson 1997) found a conformity of conclusions reached from these two aspects of Malaise trap samples taken from a range of habitats.

A co-ordinated approach would be of major benefit to ecological understanding and resource management. Given the vast extent of insect biodiversity, the numbers of workers in this area in New Zealand are pitifully few. This is particularly noticeable when compared with numbers of workers in the relatively straightforward areas of plant and vertebrate ecology. A standardised approach to insect sampling for biodiversity comparisons would therefore also provide a much needed focus for the development of taxonomic and interpretative tools in applied biodiversity studies.

Such tools include collections, purpose-built computer image-based keys and analysis packages, and integrated databases of species, autecologies and reference materials. A standardised focus could also provide guidance to autecological studies of insect species that, because of their abundance, apparently fulfil an important role in various habitats. This would give some direction to Lawton's (1993) call for taxonomic expansion of autecological studies.

6. Conclusions

6.1 INDICATORS IN THE NEW ZEALAND SITUATION

Selection of biological indicators must be guided by the uses to which they will be put. Current management needs include applicability in a wide range of different situations, and the assessment and comparison of community qualities such as characteristics of biodiversity and degree of endemism. These applications direct the need for potential indicators to be ubiquitous in terrestrial systems, and to be characteristic for the community to which they belong. Documentation of community qualities demands the use of species assemblages, as do comparisons that extend beyond the range of single species. For comparisons to be meaningful, methodology must be able to be standardised, and this is only possible for studies widely separated in space and time, when using defined protocols for documenting assemblages rather than single species. Particular species that are indicative of particular situations, or that are of particular interest may then be drawn from (or related to) this community context.

The high degree of endemism, and unique representations of taxonomic groups in the New Zealand flora and fauna, are indicative of a need to utilise methodologies which are appropriate to the temperate New Zealand systems. Many groups utilised in the Northern Hemisphere or the arid systems of Australia are inappropriate for this country.

Vegetation forms the most stable and visible indicator of terrestrial communities. These respond directly to the three direct environmental parameters of temperature, water availability and edaphic qualities, leading to the ability to define bioclimatic zones. They are also considerably influenced by biotic factors in the local environment, e.g. anthropogenic pressures, including introduced grazing mammals. A large amount of background information exists which pertains directly to the New Zealand situation.

Insects supply the vast majority of biodiversity, and their communities are sensitive to systems and processes. Insects may therefore supply interpretable information on both the status of the biodiversity of a system, and of processes occurring within these systems.

6.2 THE RECOMMENDED APPROACH: AN HIERARCHICAL MODEL

How a community is measured not only defines our perception of the community, but also influences our ability to measure community response to extrinsic or intrinsic disturbance.

An hierarchical approach to documenting of the status of the biological estate appears appropriate for all situations. This approach allows a 'focusing in' on a particular problem, and comparison, and therefore testable extrapolation, at broader spatial scales. In order for information to be comparable, some standardised protocols in documentation scales need to be followed.

The broadest scale is geographic, relates to current ecological regions and districts, and is amenable to current and future GIS capabilities for mapping general vegetation systems. The land area of New Zealand has already been subdivided into ecological regions, based on their bioclimatic attributes. Vegetation types are already recognised and used by land managers, and their mapping within the GIS environment is well advanced by several local bodies and land managers. Satellite imagery currently supplies coarse information for this level, and resolution of data from this source will improve. More detailed remote sensing techniques, such as photography and airborne video also have the ability to supply data resolution within the second level, at the local, or patch dynamic scale.

Components within systems are identified at the habitat or patch dynamic scale. An existing standardised methodology for vegetation (recce plots) allows access to a great deal of historical data. Recce plots supply semi-quantitative details on vegetation components, their structure and demography. Plots may be given marked centres for relocating and monitoring change with time. Data is amenable to multivariate analysis techniques, and a software package for this is already in existence (Allen 1992, Hall 1992).

Some minor additions are suggested to the traditional model, to allow recce plots to better reflect current information requirements. Although originally recommended for homogenous vegetation types, there is no reason why recce plots should not be used in ecotones and areas of obvious system change. Such areas are often those of particular interest to the ecologist and manager. Secondly, the relationship of plot diameter with vegetation canopy height may be relaxed, such that all plots are of similar size to those in forests (approx. 30 m diam.). This would allow comparisons at either end of a succession to forest to be more easily compared. Thirdly, supplementary information on vegetation health and debris resource should be included in recce plots, to allow relationships with insect biodiversity to be better investigated. Vegetation health can be done in a relatively pragmatic manner using fixed centre forest health plots, as used by Hosking and Hutcheson (1988) in beech forest. Debris can be categorised into size classes, distance from recce plot centres, and decay classes. A text for wood decay fungi was provided by Hood (1992), and some work is currently being conducted into log decay by Peter Buchanan, Landcare Research, Auckland (Stewart and Allen 1998).

The habitat or 'patch dynamic' level allows questions on insect community biodiversity to be addressed using Malaise-trapped beetles. This approach has been validated in several systems over some time, and was specifically developed in New Zealand systems. It is pragmatic, relates to the scale of recce plots and delivers comparable information from most systems. Information on vegetation health and debris resource should accompany insect sampling, as these parameters relate to system dynamics and the biodiversity which essentially performs the processes within systems. Documentation of insect

biodiversity, requiring these expanded details of the insect community habitat, also relates to the third level of scale.

The third level of scale relates to a narrowing of the questions being asked, to address particular species or problems. These lead to a more intense level of measurement and analysis, and accordingly, there is an expansion of appropriate methodologies, dependent on the species involved and the question being asked. Several of these are given in section 2.2. There may also be an expansion of the geographic area involved, in association with studies of single species, whether these are indicators or 'flagship' species. However, the first and second levels of scale provide a pragmatic approach to the initial system documentation and the formulation of hypotheses. Collection of more detailed information, if made supplementary to these core methods, would ease the extrapolation of the detailed study results to broader geographic scales. It would also enable cross-referencing and comparison between studies conducted by diverse workers. A short handbook of simple methods for vegetation sampling at the third level of scale is presently being compiled by one of the authors (D.R.G.).

The conclusions outlined here are guided by several factors. These include the inability of single taxa to provide information interpretable at the broader community level, or to relate to every habitat type; the complex and dynamic phenomena we recognise as communities; and the need for documentation to be relatively pragmatic. Management perceptions, the localised scale of most management operations, and the current GIS-based mapping abilities logically guide us toward an approach which is based on vegetation systems.

Watson et al. (1995), in their summary of the extensive 'Global Biodiversity Assessment', reported that currently no small subset of indicator species can be used to monitor all aspects of biodiversity relevant to management. Single indicator species cannot be used in habitats they are not present in, and such an approach leads inevitably to each problem or management question being approached using a different indicator, resulting in a plethora of separate approaches, and no ability to pool information. Standardised taxonomy is not an impediment where entities are relatively easily identified, as are plants and vertebrates, but is vital for cumulative entomological results.

A limited number of standardised sampling techniques can provide a simple core methodology across all studies and would provide baseline information about the biodiversity of various systems and their dynamics. This would allow the relative influence of management effects on biodiversity to be tested against this background knowledge.

Species that are known to be indicative of particular habitat conditions may be used to interpret factors influencing the communities they occur in. Hypotheses about such influences may be substantiated from the biology of other species both within the initial subset, and within other subsets. This 'raw biological knowledge' utilises the predictive power provided by systematic relationships within the biota. Its use in the interpretation of system influences is often undervalued in favour of statistically significant numerical correlations. It should be more widely understood that data derived from knowledge of component species biology (e.g. the trophic habits of insect species), provides 'real' information on connections within systems which are non-statistical and

from which conclusions may be drawn which are more powerful than simple correlations.

Interpretation of relationships within systems may be enhanced by studies of those species that are most common in communities. Although these are obviously of functional importance in systems, they have traditionally been ignored in conservation ecology. Their occurrence as pests in production systems is often not documented within a community context, and there have been few comparisons of such widely varying systems to date. This is to the detriment of informed sustainable management of both conservation and production systems.

The hierarchical model, based on standardised protocols for documenting and comparing communities, would enable such comparisons. Hypotheses about influences on biota may be erected from observations at level two, and tested by properly designed experimental approaches at level three. These may use appropriately stratified sample groupings of previously collected data, or more intensive sampling approaches in new studies.

The recommended approach uses relatively pragmatic standardised and validated methodologies, and does not preclude the linkage of 'non core' taxonomic groups or methodologies to the core data, but rather offers a broader system context for other groups and studies. This is particularly relevant to the insect sampling, where protocols provide standardised samples from a wide range of orders.

6.3 THE NEED FOR STANDARDISATION

As any monitoring obviously requires comparison of systems over time and space, it is imperative that a defensible sampling approach is taken, and that methods of sampling, taxonomy and comparison are standardised (e.g. Hutcheson and Hosking 1994, Ehrlich 1996, Hutcheson 1997). Information from various studies may then be compared, using the standardised methodology to supply a 'common language'. The model for this is of course systematic taxonomy, which forms the basis of our present understanding of the biosphere.

National standardisation of methodologies could save an enormous amount of resources, but such standardisation may be difficult to implement within the current competitive science structure, as every group will wish to control a centralised single database. We are hopeful that this situation will mature and that the needs of the end-users will take precedence, so this should not be seen as a reason not to proceed. In the interim, collaboration between research groups to ensure a standardised approach to methodologies (particularly of insect taxonomy) is absolutely crucial. Data from separate studies may still be kept separate (to conform to New Zealand's present fragmented science structure) but interpretations and conclusions may be shared.

It is noteworthy in this context that the majority of New Zealand's entomological expertise that could be applied in this area now lies outside the CRIs. However, funding agencies have the power to audit research programmes, to ensure that the collaborations necessary to deliver appropriate answers for both management and science are procured.

7. Acknowledgements

The authors acknowledge the New Zealand Department of Conservation for funding this work under consultancy agreement No. 2242, and are grateful for the advisory role of Greg Sherley of DOC Science and Research.

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Appendices

APPENDIX 1. SELECTED EXAMPLES OF VEGETATION MONITORING

A1.1 Hierarchical investigations into *Nothofagus* spp. forest health: New Zealand: 1982–95

References: Hosking & Hutcheson (1986, 1988, 1999), Hosking et al. (1990, 1993).

Background

Beech forest health declined over extensive areas in the late 1970s, and a hierarchical approach was taken to investigating the causal mechanisms.

Brief description of methodology/approach

Investigations began with a survey of where problems had been observed by field staff, followed by visits and observations in several areas. A sequential series of study sites was established in a subset of areas, and several heuristic methodologies from broader to more focussed studies successively employed to evaluate possible causes. These included: non-permanent tree centred forest health plots (FHPs), permanent FHPs and permanent plots with fixed canopy access. Data was collected on insect and pathogen abundance, litter fall, meteorological information, ground moisture, tree physiological parameters and tree growth records from increment cores.

Summary of results

We concluded that we were observing system dynamics responding to environmental forcing. Forest health in different areas varied in accordance with the demographic structure of the forest, local site conditions, and the organisms present. The general patterns revealed were found to fit Manion's (1981) model for tree health decline. This involves three factors operating. (1) Predisposing factors, e.g. stand age, or moisture retention capability of the soil; (2) inciting factors (i.e. changes in the growing rules—such as drought or wind events); and (3) contributing factors, e.g. insects and diseases. It became clear that plant physiology varies with environmental change, and that insects are sensitive to this. The long-term influence of wind events, and exposure over time on beech forest was also demonstrated.

Indicator value

Foresters observed changes in herbivory associated with insects, and tended to blame the proximate agent, although these were simply responding to (and therefore indicative of) changes associated with system dynamics. Older trees showed less resilience to environmental change and increases in herbivorous insects and pathogens were indicative of this decreased plant vigour.

Detritivores associated with debris breakdown are of course also indicative of resources available. There is a long gradient between live and dead plants, and colonisation by insects and fungi occurs over this whole gradient (Hutcheson 1996) although a great deal more study of these processes and interactions is required.

Other lessons

It was necessary to assess the systems as a whole (i.e. plants, insects and pathogens) to understand the processes occurring within the local communities. The insects were integrating the extrinsic environmental factors, the intrinsic edaphic factors and the changing status of the vegetation systems. Interpretation of system processes can thus be available from the biology of species in insect communities, together with appropriate documentation of the vegetation systems.

The use of a hierarchical approach allowed the construction of a system context within which to refine questions and to appropriately view results from the smaller scale more intense studies. Knowledge of the biology of species observed in the smaller scale level 3 studies, together with the broader context provided from higher level 1 and 2 studies, enabled the interpretation of a range of local system responses within a single understandable model. Simple, tree centred, forest health plots (FHPs) were found most useful, where individual tree health was recorded as healthy, intermediate, unhealthy, dead or windblown. These were permanently marked and allowed rapid re-evaluation of changes in forest health.

A1.2 Hierarchical investigations into pohutukawa health: New Zealand: 1982–95

References: Forest Research Institute (1989), Hosking & Hutcheson (1993).

Background

Anecdotal reports of declining pohutukawa health implicated a range of agents including environment, possums and insects. A hierarchical approach to investigating the causal mechanisms was taken.

Brief description of methodology/approach

Investigations began with an aerial coastline survey of the northern North Island over the complete endemic range of the species. This was followed with rapid demographic studies and assessments on 197 sites. This in turn was followed by detailed study of trees from a selected site covering a range of health conditions. These were assessed every 2 weeks over 2 years, with documentation of tree phenology, damage and damaging agent. Binocular assessment of canopies was made, allowing determination and categorisation of insect and possum damage. Litter traps were used to collect fallen foliage, which was also separated into damage classes and categories. In addition, growing twigs and buds were marked and sequentially assessed.

Summary of results

The initial (level 1) survey indicated that most of the area previously occupied by pohutukawa had been cleared for farmland. The second level study showed

that some insect problems were present in some areas, however that possums appeared to be the major damaging agent of current vegetation, and that their effects appeared greater on more mature trees. This level 2 study also showed that regeneration was only present on 15% of the sites, and that these were sites from which domestic stock were excluded. Neither goats nor possums would have been prevented by the stock fencing, present from accessing the sites that did have regeneration. The third level of study showed that extending buds were very palatable to possums, whereas once expanded, the fresh foliage was relatively unpalatable. Older, hardened off foliage was again palatable. A mature tree only holds the most recent 1 or 2 flushes of leaves, and continual loss of the new buds severely reduced photosynthetic production, and led to a redirection of resources from flowering to replacement of foliage buds. Possums continually removed the replacement buds on favoured trees, leading to death of mature trees in a very short time. Trees in a more juvenile growth mode were more vigorous, producing longer growth with each flush and retaining more leaf flushes. They were therefore able to withstand possum depredations to a greater extent and over a longer period.

Indicator value

An interesting conclusion from this programme was that lack of something is unable to be noted unless it is set within a larger context. Hence the lack of pohutukawa forest showed up from the level 1 study, which was able to assess where we could have expected pohutukawa to have grown. The lack of regeneration showed up with the extensive demographic comparisons in the level 2 study. The removal of extending buds by possums only became apparent from the level 3 study, when buds were individually marked and their development followed.

It could be said that: farmland is indicative that the previous forest cover is gone; that presence of domestic animals indicates that no palatable indigenous plants will be able to regenerate to replace this forest; and that high possum numbers are indicative of the imminent demise of extant pohutukawa trees. However such interpretive conclusions are a consequence of the hierarchical study which places each series of observations within a system context.

Other lessons

Managers were able to use this knowledge immediately to target possum control application just prior to spring bud development. This allowed trees immediate release and regrowth, which, after a single season, enabled them to begin flowering again. In addition, the information from the first and second levels on the diminished extent of this favoured tree, and the lack of regeneration occurring due to domestic stock, underpinned widespread involvement of the public through Project Crimson. This has had a major effect on not only the enhancement of the pohutukawa round New Zealand, but also on the retention and expansion of our much depleted coastal forests through the fencing out of domestic stock in many areas.

Once again, the use of a hierarchical approach allowed the construction of a system context within which to refine questions, and to appropriately view results from the smaller scale, more intense studies. Accurate observation of processes at all scales enabled a range of appropriate management responses to be initiated.

A1.3 Hierarchical monitoring at Tiger Creek Preserve; USA; 1989–95

Reference: Menges and Gordon (1996).

Background

A three-level hierarchical monitoring scheme ranging from more general and superficial (level 1) to more intense monitoring (level 3) was applied to narrowly endemic plant species in a Florida nature preserve. The authors discuss the scheme for two species: *Nolina brittoniana* and *Warea carteri*.

Brief description of methodology/approach

For *Nolina* all known populations were mapped (level 1), population size was determined for 14 arbitrarily selected populations (level 2), and detailed monitoring of rosette dynamics was done for 5 populations (level 3). The level 3 monitoring involved sampling within 10 x 10 m grids randomly located within a population and mapping and tagging of individual plants. Leaf number and length was recorded, along with reproductive information.

For *Warea* (which is an annual species in communities dominated by perennials) level 1 information included appearance and disappearance of all known populations and relating this to fire history. Level 2 monitoring expanded the interpretation of post-fire population dynamics using complete counts of all known populations. Level 3 monitoring was not done but is clearly needed: seed dynamics in relation to patterns and mechanism of population size change, colonisation and loss.

Summary of results

For *Nolina* level 1 data was rather inconclusive, partly because it became apparent that not all populations were discovered. The main conclusion at level 2 is that populations are generally expanding although clearly longer-term data sets are needed. Level 3 provided useful information especially on dynamics of populations and the habitat; although population growth estimates are preliminary it was possible to calculate an expected population growth rate and thus provide information for predictive management.

For *Warea* it was apparent at all levels that populations are dynamic and (in some cases) ephemeral. It seems likely that fire triggers germination of dormant seed and leads also to high seedling survival, so that continued fire management is suggested. However, seed bank data required a more intensive level 3 approach to achieve good predictive management and to use the species as an indicator of habitat condition.

Indicator value

At a local level, this project demonstrates the value of even rare species for specific management of a particular site. The three-level scheme can give a logical approach which optimises the usefulness of information gained in relation to expenditure of resources. What this work also shows is that, for an indicator to be of greatest use, we have to not only ask the right questions but also investigate and measure it at an appropriate level of intensity.

Other lessons

What Menges and Gordon have done is to put more precision on what is often done on a more 'seat of the pants' intuitive basis—that is, to select subsets of sample sites for more intense investigation and measurement. Although their framework is extremely simple it can lead to a more rational allocation of resources for use of indicator species, so that not only the right indicators are selected, but those that are will be investigated at an appropriate intensity of effort.

A1.4 Hierarchical approach to measuring fire frequency effects; USA; 1938–90

Reference: Faber-Langendoen and Davis (1995).

Background

Prescribed burns have been used as a means of managing oak barrens since 1962, thereby controlling the environment using deliberate management tools. To evaluate fire management success a reasonably robust method was needed which had potential to give a historic picture.

Brief description of methodology/approach

Aerial photographs from 1938 and 1960 (prior to burning) and from 1987 (25 years after the burning started) were digitised with change in tree canopy cover being compared. More detailed information was obtained from three 50 x 75 m plots established in 1990 positioned so as to include different management regimes. All trees >10 cm dbh were measured for diameter and 18 randomly selected trees from burnt and unburned sites were bored by increment corer. These plots were located on the aerial photographs.

Summary of results

The current management regime of burns every 4 years may not lead to the desired end-result which probably requires a shorter fire-return period. It was also found that fire had somewhat different effects at different spatial scales, rather complicating both the management needed and also the way in which it should be interpreted.

Indicator value

This project is a further example of the kind of tiered procedure advocated by Menges and Gordon (1996). It provides an example where there is clear need to use the indicator species (in this instance canopy trees) at a range of scales, and also to use these indicators on a regular basis as a means of assessing the effectiveness of a particular course of management. In particular there is need in this instance to monitor on a larger spatial scale than is currently the practice.

A1.5 Hierarchical approach to the use of wetland macrophytes as indicators; USA; 1938–87

Reference: Wilcox (1995).

Background

Hydrologic disturbance is widespread and affects wetland and aquatic macrophytes by creating temporal changes in soil moisture and water depth. These changes can be natural and help maintain wetland diversity or can have negative effects (especially when human induced). Macrophytes (the obvious plants associated with wetlands) often respond very quickly to environmental changes such as water level, oscillations in water level and water quality.

Brief description of methodology/approach

Aerial photographs for the study site were available for 19 years over a period ranging from 1938 to 1987. Percentage of wetland was estimated from each photograph along with vegetation maps. The vegetation maps were ground-truthed resulting in eleven prominent vegetation types. Using a grid map, ten 1 x 1 m plots were placed randomly in each vegetation type and sampled for species present and for estimates of percentage cover. Water level data was recorded at bi-weekly intervals over a 2-year period.

Summary of results

From this approach the hydrologic history of the site could be reconstructed through various anthropogenic disturbances. The photographs provided essential 'moment-in-time' information on vegetation patterns, type and abundance. From this it is possible to develop a management regime which will lead to reasonably predictable effects on vegetation and component species.

Indicator value

In this example each vegetation type acts as an indicator species and the value is in the relative proportions of each and the patterns which they form over time. Indicators at this level of spatial scale can be interrogated by remote sensing, but still require ground-truth data to verify the interpretation of aerial photographs.

Other lessons

This is one of the simpler and perhaps better examples of the use of remote sensing to evaluate indicators. Wetlands are probably particularly amenable to this kind of approach.

A1.6 Long-term monitoring of orchids; United Kingdom; 1966–79

Reference: Wells (1981).

Background

Interest in orchids and their biology and especially in the population fluctuations which may occur over a number of years is increasing. They are also often useful indicator species. Therefore it was relevant to examine the information which might be obtained by regular monitoring. For this purpose

Wells initiated long-term studies of populations of *Spiranthes spiralis*, *Acerus anthropophorum* and *Herminium monorchis* growing in chalk grassland. These studies extended over a period of 14 years.

Brief description of methodology/approach

Plants were recorded using a method which enabled the position of individual plants in relation to permanent marker pegs to be plotted. The method uses a long transect line which is laid out between two permanent marker pegs which can be subsequently relocated. Two measuring tapes are zeroed at two predetermined points along this line (e.g., at the 0 m and 5 m marks) and are then run out to the plant being recorded, in what is essentially a process of triangulation. The position of the plant is thus a combination of four essential pieces of information: the position on the transect tape at which tape A is zeroed and the distance along tape A to the plant, and the equivalent information for tape B. Using this method plants can be relocated with an accuracy of about 1 cm. This means that it is simple to return year after year to the same plant (or to its position when dormant). Data recorded for the target species was: (a) state of plant—flowering or vegetative, (b) height of inflorescence above ground level, and (c) number of flowers. By measuring the same plants year after year it was possible to construct life-table analyses and to follow the fate of individual plants as well as monitoring the appearance of new plants; it was also possible to determine when plants were dormant in any particular year.

Summary of results

For each species it was possible to determine average and extreme mortality rates as well as longer-term trends in mortality (and a measure of 'health') by number of flowers and inflorescence height. This could be correlated with changes in management. For instance at one site there was light cattle grazing with intermittent rabbit browse, with an increase in *Bromus erectus* pasture until a cutting regime was introduced. Management could be correlated with rates of recruitment of new individuals to the population. It was possible to determine by this method that each species has unique characteristics in terms of recruitment and mortality and in fluctuations in population size. A very important conclusion was that not only flowering varied greatly with not all plants flowering in any one year, but that in some years some plants did not appear above ground at all. Without the sampling regime used, this would have been unnoticed.

Indicator value

Orchids tend to be very site specific, not only in terms of associated vegetation, soil moisture and sometimes specific species associations, but also in terms of essential mycorrhizal fungi. The relatively high turnover rate of orchids makes them especially useful as indicator species although it will be necessary to do annual censuses for several years in order to obtain useful data.

Other lessons

This method has been also used in scrub and shrubland for location of plants of *Hebe armstrongii* in Canterbury (D.R. Given and A. Baird unpublished data). Under these circumstances with two people handling the tapes it proved to be a

very fast way of both mapping and relocating plants along a transect up to at least 12 m wide. Its advantage is that it rapidly provides a means of locating and measuring particular individuals without having to resort to individual markers beside each plant.

A1.8 Mortality over a 20 year period for trees; USA; 1966–85

Reference: McCune and Henkel (1993).

Background

A particularly valuable indicator of environmental change and condition is the rate of tree mortality and long-term trends in mortality change. This has been mooted as especially valuable for assessing the effects of air pollution. Further, tree mortality drives forest dynamics. Permanent plot data have no perfect substitute in plant ecology, but these data accumulate only slowly and there has been little success in applying them to address policy and management issues. A regional database of tree mortality allows evaluation of possible causes for spatial and temporal variability in mortality of natural forests. The study area is in high diversity transitional forests of western Kentucky.

Brief description of methodology/approach

Living and dead trees were sampled in 40 randomly placed circular plots arranged along transects. Living trees >4 cm dbh were measured and identified in 100 m² plots; species, year of death and size of each dead tree were determined in 300 m² plots using the same plot centres as for the living trees. Cause and year of death was estimated using 'tree condition' (factors such as bole breaks, whether twigs were intact, degree of trunk rot etc.). Older deaths were dated using increment coring on sapling, which had grown up in the light gap.

Summary of results

There had been earlier observation on the sample sites (1930s) and it was possible to observe a shift in species dominance. Mortality rates were able to be established at each site and were comparable to those which have been calculated for other midwestern US forests. Most mortality was in oaks and while at least 28% was attributable to wind at one site, only 3% was similarly wind related at the second site. There was a slight increase in mortality in the second decade of observation.

Indicator value

This kind of data gathering is relatively simple but effective in assessing rate of loss of species which are functionally keystones. It might be more difficult to use in higher rainfall areas of New Zealand where decay rates are probably rapid, and allowance in beech forests would have to be made for arrested growth at the juvenile stage in *Nothofagus*.

Other lessons

Comparable studies have been undertaken elsewhere in North America. One of the most interesting other studies is, however, one which serves as a cautionary lesson. Szafoni et al. (1994) tested the idea that large trees are an indication of

past vegetation and ecosystem condition. They took increment cores from seven large-diameter (85–160 cm), open-grown bur oaks in a central Illinois prairie grove, where it was assumed that the trees were relicts from past savanna. However, the results indicated that growth rates for open-site trees of this species were much faster than growth rates for trees in a closed-canopy site and that all the trees were ‘post-settlement’ in age. Therefore the assumption that such large trees were relicts from pre-settlement times was unfounded.

A1.8 Trampling of heathland; USA; 1987–90

Reference: Sutter et al. (1993).

Background

The effects of recreation use including trampling by people is an increasing concern in maintenance of ecological integrity and biodiversity. It is of considerable concern in New Zealand protected areas, especially National Parks, and is most aggravated where there are regular sporting events such as the Arthurs Pass ‘Coast to Coast’ race. Most studies of the effects of trampling have used one of two study designs: (1) comparison of vegetation along an established trail with vegetation some distance from the trail - usually at one point in time, or (2) simulation of trampling at various controlled levels and subsequent comparison of rates of recovery between different levels of trampling intensity and between different community types.

Brief description of methodology/approach

The approach here was to evaluate the effectiveness of a boardwalk at protecting a rare plant community by measuring the impact of hikers leaving the board walk and crossing over intervening vegetation over a 4-year period. Twenty-one belt transects were randomly placed within low heath spanned by the board walk and six further transects preferentially placed in areas predicted to receive intensive trampling (usually because of previous use). The transects started from the board walk and were perpendicular to it.

Permanent photographic plots 1 m in length and 0.5 m wide were established on each transect where vegetation was less than 1 m high. Plots were photographed in August of each year, and the photographs analysed by projecting them onto a rectangular grid of 200 points, with cover data scored by counting hits on each species and dividing by two to give a percentage. Data were collected at three levels of intensity: (1) presence/absence of trampling, (2) severity of trampling using four classes, (3) % cover of key plant species and of exposed soil and rock were calculated. Control plots were not established because it was not possible to predict which areas would be used by walkers and which would not.

Summary of results

The number of trampled plots increased from 54% to 64% during the experimental period. Changes in cover of sand myrtle and of exposed rock were significant and indicated a key ‘reaction’ to the effects of trampling. The three data levels provided different levels of information. Evidence of trampling was seen in the simplest level of data input (presence/absence); the second level gave a better picture of how trampling damage was progressing over the course

of the study; the most detailed level of analysis showed specific decline in sand myrtle and increase in exposed rock.

Indicator value

This study showed that one particular shrub species was a good indicator of trampling (sand myrtle) especially in combination with measurement of bare ground. This means that future studies can concentrate on these parameters allowing sampling of more sites and a greater number of plots. Thus there may have to be a rapid level 2 pilot study first to determine what species are the good indicators.

Other lessons

This kind of problem is a common one on conservation lands and this research suggests that there is room to develop additional experimental techniques as well as a vulnerability index (Appendix 2) for trampling of plant species so that a small number of good indicator species can be identified.

A1.9 Assessment of plant biodiversity change over time in plots; USA; 1941–89

Reference: Schlesinger et al. (1994).

Background

This project also incorporates a degree of hierarchical inquiry. It was set up to determine if composition biodiversity (measured using plant species) was changing over time within a research natural area. As well as using data from plots established in 1978, earlier data sets were used as well to provide a longer data set.

Brief description of methodology/approach

For 153 plots set up in 1978 (nested 0.1 ha and 0.01 ha plots) all trees >15 cm dbh were measured and mapped. In addition data was used from three other surveys of the study area: 1941 (50 plots, 0.01 ha), 1963 (2 plots, 3.31 ha and 1.94 ha), and 1972 (40 plots, 0.025 ha). Three indices of biological diversity were used: species richness, heterogeneity (Shannon-Weaver index), and equitability (Pielou's index). Variance estimates were computed for each subsample of the plot data from 1978 to 1989 and corresponding to the respective surveys from 1941, 1963 and 1972—these were used to test for whether biological diversity had significantly changed during the general time periods.

Summary of results

Permanent plot data was found to provide the most reliable comparisons of changes in biodiversity through time. One of the earlier data sets did not seem to represent the stand or sample area as a whole but when this set was excluded reasonable trends could be established. Diversity decreased slightly during the 11-year period of the permanent plots but that prior to this there had been some increase in diversity.

Indicator value

This project suggests that best value for indicator species may be obtained through use of well-positioned permanent plots rather than through plotless sampling, and that if plots are not carefully positioned they may produce anomalous results. Indicator species can be not just single species but whole suites of species using appropriate indices of overall 'biodiversity' (but see also discussion in section 5.6.1).

A1.10 *Muehlenbeckia astonii*: status and occurrence; New Zealand; 1990–97

Reference: D.R. Given (unpublished data).

Background

This endangered species of the family Polygonaceae, sometimes known as shrubby tororaro, has a distribution which includes four discrete areas: coastal Wellington, north-eastern Marlborough, north Canterbury and Kaitorete Spit just south of Christchurch. The study above is concerned only with the North Canterbury and Marlborough populations. Here the species is reduced to fragments, often only single plants, and reproduction is virtually unknown.

Brief description of methodology/approach

Essentially a level 1 survey with some level 2 features, for all plants seen, GPS data was recorded (latitude and longitude to 0.001 of a minute). Plants were each given a unique field number and individually photographed. Maximum width and height was recorded and where possible plants were sexed. Soil type, slope, aspect and surrounding vegetation was noted.

Summary of results

The species has a surprisingly restricted niche but tends to persist for a long time, although usually with no effective reproduction occurring. It is in decline, and where single plants or small groups remain it has usually lost most of its original associates, being now often found on grassed terrace faces which lack other scrub species. The survey led to discovery of a significant number of previously unknown plants.

Indicator value

Although such a rare plant, it has potential indicator value because of its longevity (over 100 years) and its wide distribution. Indeed it already indicates the severe decline in indigenous cover in drier downlands of the eastern South Island. It is probably an ideal plant on which to carry out a hierarchy of monitoring along the lines suggested by Menges and Gordon (1996).

Other lessons

This species belongs to an ecological group of species with somewhat similar ecology and distribution, but with varying degrees of abundance, leading to the possibility that they could collectively form a guild of indicator species.

A1.11 *Leptinella nana* population study; New Zealand; 1989–96

Reference. Given, D.R. and A. Baird (Unpublished data).

Background

Leptinella nana (Asteraceae, tribe Anthemidae) is a highly endangered endemic herb known from only three populations in New Zealand; Titahi Bay, Rai Valley, Port Hills (Christchurch). At each site populations of *L. nana* cover very small areas, overall amounting to only a few square metres within which distribution is discontinuous to sparse. It characteristically occurs on small patches of bare soil where seeds germinate during periods of cool, moist weather. Monitoring of one site over a 5-year period shows that there is high variation in abundance from year to year. Requirements for micro-site disturbance appears to have led to a strategy in which there is continual dying out and re-establishment of plants.

Brief description of methodology/approach

The Port Hills site has been monitored between June 1989 and August 1996. Here the percentage cover of *L. nana* has been recorded for 500 mm² plots along five transect lines each plot at an interval of one metre so that plots alternate with non-plots. This gave a total of 42 individual plots with a total area of 10.5 m². Estimates of population size were made by using a string grid with strings 5 cm apart. This gave a grid within the transect of 25 squares, each representing 4% of the total plot area. Cover was estimated by eye—estimating the number of complete squares (and fractions of squares) overlying the target species.

Summary of results

Considering all plots, mean percentage coverage variation from 1989 to 1993 was as shown in the table alongside. Some plots, generally those with very low frequency of *L. nana*, show little variation; in others there is very significant yearly variation in cover, mostly reflecting establishment and extirpation of individual plants. The species appears to have a pattern of continual extirpation at a 'micro'-level with (at least in good years) replacement by new recruits.

YEAR	% COVER
1989	0.62
1990	1.92
1991	3.07
1992	1.45
1993	2.45

Indicator value

One implication of this monitoring experiment is that it suggests that abundance should be monitored at a range of scales: geographic (number of sites), spatial (number of plants or area in each population), temporal (change in abundance and pattern of occurrence over time). The experiment also indicates that:

- (1) it is not necessary to count individual plants (indeed this would be very difficult with this species) but that an estimate of cover can suffice;
- (2) that cover estimated by use of a gridded plot can be done quickly but is reasonably robust and consistent from year to year; and that
- (3) combining results from a number of replicated plots can give quick estimates of overall population trends.

Other lessons

In this particular instance the data—although involving a very local and rare species—is of site value in assessing environmental trends because the study area lies along a popular walkway. The *Leptinella* has a very vulnerable niche; it appears to like a small amount of disturbance because this creates bare soil surfaces available for colonisation; on the other hand too much disturbance (especially trampling) and the patches of plant are killed.

The natural disturbance regimes to which *L. nana* is adapted are highly modified by anthropogenic influences. Monitoring of the Port Hills indicates that the present level of disturbance at this site, chiefly through human tracking, is probably sufficient but that a small increase in usage would probably lead to decline in this species.

APPENDIX 2. EXAMPLE OF THE ‘VULNERABILITY INDEX’

VULNERABILITY INDEX (REFERRED TO IN SECTION 3.5.2) APPLIED TO *CORTADERIA TURBARIA*, OVERALL VULNERABILITY INDEX SCORE = +1 (SOURCE: GIVEN 1994).

FIELD	HIGHER CONSERVATION PRIORITY	LOWER CONSERVATION PRIORITY
Geographic	a. Small range	Wide range
	b. Endemic to region	Not endemic to region
Taxonomy	a. High level taxon*	Low level taxon*
	b. Genus/family small	Genus/family large
	c. Probably relict	Not a relict
Habitat	a. Under threat	Not under threat
	b. Fragile	Resistant
	c. Specificity narrow	Wide habitat range
	d. Successional	Climax
Life form	a. Annual/short lived Perennial	Long lived perennial
Populations	a. Small	Large
	b. Few	Many
Biology	a. Rarely flowering	Often flowering
	b. Specific pollinator	Non-specific pollinator
	c. Dioecious	Monoecious
	d. Chiefly outcrossing	Selfing readily
	e. Seed short-lived	Seed long-lived
	f. Specific disperser	Non-specific disperser
	g. Class structure poor*	Class structure good*
	h. Vegetation Reproduction poor	Vegetation reproduction Good
Miscellaneous	a. Harvested	Not harvested
	b. Predation high	Predation low
	c. Region of high endemism	Region of low Endemism
	d. Not on protected site	On protected site

*Notes:

Taxonomy: high level = family, genus, etc.; low level = subspecies, variety, etc.

Biology: poor = disproportionate representation of one age class.

Scoring: left column = +1; right column = -1.