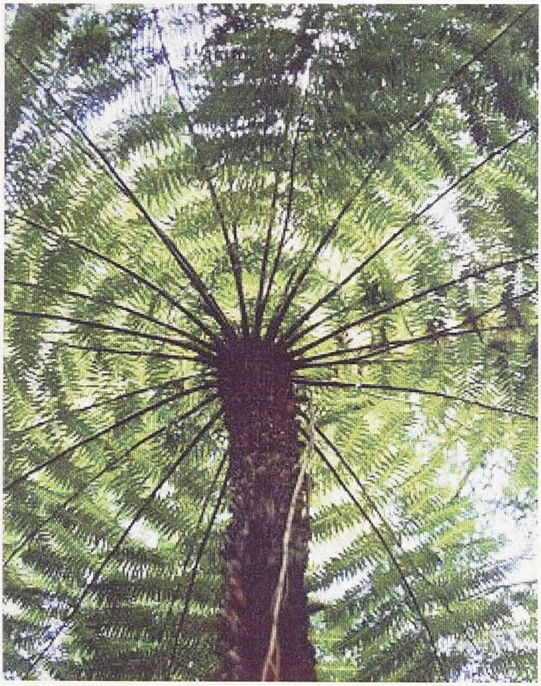
Systematic Techniques to Locate Reserves for Biodiversity Conservation

Including a case study on the conservation of floristic diversity in East Gippsland



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Declaration

Except where otherwise stated, all of the work presented in this thesis is my own. All sources used have been acknowledged. This applies also to maps used in this thesis. Except where otherwise acknowledged, all the maps were produced by me using the data and analyses described in the methods section, with the aid of GIS software.

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Abstract

Changes in land use, and resulting fragmentation and loss of habitat, are leading to significant loss of biodiversity. Maintaining systems of reserves for *in-situ* conservation is a key strategy to stem biodiversity loss, but establishing reserves is often a difficult social and political process involving decisions between a range of alternative priorities for land use. Given limitations on availability of land for reservation, and the urgent need to establish reserves in many areas, it is important that reserve proposals are based on a systematic analysis to identify places which will contribute most to long term maintenance of biodiversity.

The prevailing approach to this problem is to identify a mappable surrogate for biodiversity, and then select areas which are representative of the elements of variation in that surrogate. The IUCN recommends that 10% of each such element be protected.

This paper examines the use of surrogates in reserve selection, and describes the development of systematic, computer-supported algorithms which use ecological data to identify areas for a representative reserve system. In Australia, the current surrogate of choice for reserve design is the vegetation class. It is used in a variety of forms, which generally combine information on floristics, dominant species and physical environmental regimes to define and map classes. While this approach is widely used, there has been little published evaluation of how well a representative sample of vegetation classes encompasses the underlying biodiversity that the reserves aim to protect.

For this paper, a desktop study was performed, using environmental data for a region of East Gippsland, Victoria. Potential reserve areas were selected to create a reserve system to meet the IUCN target of 10% representation for each vegetation class in the region. A range of reserve options which met this target were generated, and then evaluated against a large flora survey dataset to assess how well these reserves would encompass the floristic diversity of the region. A simple indicator of adjacency was also used, so that areas selected to meet the targets for representation would do so while minimising the degree of fragmentation of the reserve system.

In terms of floristic diversity known from sampled locations, the representative samples of vegetation classes were only moderately representative of the plant taxa in the region.

The reserves on average included records of 70% of the region's vascular plants, and just 43% of rare or threatened plants. This analysis has limitations which are discussed in the paper.

A second stage of analysis used the flora survey data to increase the known floristic diversity of the reserves by locating reserves where required species were known to occur. With this fine grain approach it was possible to reserve known locations of all plant taxa in the study without increasing the total reserve area. The resulting reserves were small and highly fragmented. For this reason, such an approach would not be viable without either increasing the total reserve area, or carefully managing the intervening, non-reserved matrix to buffer and link the scattered reserves.

Options for refinement of the experimental method used in this study are discussed.

Abbreviations used in this thesis

AHC	Australian Heritage Commission
ANU	Australian National University
API	Aerial photograph interpretation
BLM	Boundary length multiplier
BRS	Bureau of Resource Sciences
CALM WA	Department of Conservation and Land Management (Western Australia)
CRA	Comprehensive Regional Assessment
CRES	Centre for Resource and Environment Studies – Australian National University
DNRE	Department of Natural Resources and Environment – Victoria
EVC	Ecological Vegetation Class
GIS	Geographic Information System
GPS	Global positioning system
IUCN	World Conservation Union (formerly International Union of the Conservation of Nature)
JANIS	Joint ANZECC / MCFFA National Forest Policy Statement Implementation Sub-committee
PU	Planning unit
PVA	Population viability assessment
RFA	Regional Forest Agreement
SLOSS	Single large or several small (reserves) ?
VROT	Victorian Rare or Threatened species

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1 Introduction

This paper is about which areas should be included in a conservation reserve system. While this question spans the social, economic, and political domains, this paper considers just one aspect of the question: scientific techniques which can be used to systematically identify a limited set of potential reserve areas which would be representative of the biodiversity of the area in question.

1.1 The reserve selection problem

Recent years have seen a great increase in the concern and effort directed towards the conservation of biological diversity. This is reflected in international agreements such as the Convention on International Trade in Endangered Species and the Convention on Biological Diversity, in government policies such as the National Strategy for the Conservation of Australia's Biological Diversity, and in widespread community action ranging from protesting to planting and fencing.

A core element of any strategy for biodiversity conservation is the maintenance of a system of protected areas for *in-situ* conservation. Protected areas cover six times as much of Australia as they did 30 years ago (Creswell and Thomas, 1997), and are still growing, under the direction of government policies such as the Regional Forest Agreement (RFA) process, and the National Reserve System program.

The selection of areas to establish or expand reserves is made difficult by several factors. Firstly, biological diversity is very difficult to map. While biodiversity is a powerful integrating concept to focus attention on human interaction with a living world, it is far too general a concept to be mapped or counted. Where a suitably bounded aspect of that diversity is identified for study, such as a taxonomic grouping, the collection of field data to describe its spatial distribution is an expensive and potentially unlimited task. Further, once such information is collected, its complexity may be challenging to resolve, in order to draw out mappable entities and useful guidance for conservation planning.

Secondly, even if we know what is where, the question of which areas to protect, and how to manage them remains problematic. This is because different elements of biodiversity each have distinct, overlapping distributions, and respond to different factors in the environment at different spatial and temporal scales. What is best for one species or community is unlikely to be best for all. In many cases, their life cycles and habitat needs are only poorly understood. Identifying suitable areas to support these species or communities for the long term, and determining appropriate management for those areas, is thus difficult.

Thirdly, even where the best areas for *in-situ* conservation are identified, they may not be available, because much of the land which could be set aside as protected areas is used or valued for other purposes. The complex social, economic and cultural issues surrounding land-use mean that establishment of new protected areas must be carefully justified. To receive support, such proposals are usually closely limited to areas shown to be essential for conservation or not valued for other uses.

Bringing these three points together, the nub of the reserve selection problem is that despite a lack of certainty about the distribution of elements of biodiversity, and in many cases of their conservation requirements, reserves must be selected within tight constraints created by other land uses.

The definition of the problem in these stark terms is, however, relatively new. Biodiversity conservation has not always been the main factor guiding the siting and establishment of reserves. In order to understand the current context for decisions about reserve selection, it is thus useful to consider where current reserves and the expectations surrounding them, have come from.

1.2 Reserves and biodiversity conservation – recent partners

1.2.1 Protecting 'worthless lands' – the American precedent

The first national parks, in the modern sense of a state managed area set aside in a natural state, were Yosemite, and then Yellowstone, established in the United States in the 1860s and '70s. According to Runte (1979), their purpose was to set aside great scenic wonders of the natural world, as monuments to display the grandeur of a 'new' country which could not match the cathedrals, castles and other cultural treasures of Europe. Ecology (or equivalent terms of the day) were not mentioned in advocating the new reserves. The large area covered by Yellowstone National Park, Runte argues, reflects uncertainty about the location of undiscovered scenic wonders, rather than a desire to protect wilderness or maintain the integrity of its ecology.

Runte also proposed that although spectacular scenery and consequent tourism potential were the primary reasons for the early US nature reserves, the fundamental criterion for acceptance of a reserve proposal was that the area was essentially worthless for all other economic uses. Addressing the US Senate to argue for the creation of Yellowstone, John Conness began:

> I will state to the Senate that this bill proposes to make a grant of certain properties located in the State of California, that are for all public purposes worthless, but which constitute, perhaps, some of the greatest wonders of the world. (Conness, 1864 cited in Runte, 1979, pp.48-49)

The worthlessness of Yellowstone was repeatedly affirmed and carefully justified by those advocating its reservation (Runte, 1979). The association of national parks to worthless lands set the tone for future reserves, and was the basis for the excision of areas which were subsequently found to have value for resource uses.

1.2.2 Australia – a similar path

Australia soon took a similar path, establishing Royal National Park near Sydney in 1879 (Whitehouse, 1990). The trend of reserving scenic areas and favouring land not valued for agriculture or other resource uses, is well documented in Australia (Kirkpatrick, 1987; Pressey, 1994). Although the stated reasons for park establishment were based on local scenery, protection of productive resources or the promotion of local tourism (Whitehouse, 1990), areas which over the years have been dedicated to nature conservation, are mainly of little other economic use. Where an economic use has been later found, such as forestry, mining, hydro-electricity or intensive tourism, this has led in various cases to the revocation of reserve areas (Mercer and Peterson, 1986).

These factors in decisions about land use have created a particular pattern in land tenures. For example, in eastern Australia, forests set aside for timber production have tended to be on lands originally deemed unsuitable for crops or livestock (Pressey, 1994). This bias toward less productive areas is even stronger for land dedicated to conservation (Mercer and Peterson, 1986; Kirkpatrick, 1987; Braithwaite et al., 1993; Pressey et al., 1996a).

1.2.3 Reserves as a system to conserve and 'represent' biodiversity.

Since WWII there has been a great change in the purpose and value attributed to nature reserves. Nature-based tourism and recreation are still important, but the conservation of natural ecosystems and native species have increasingly become the main factors driving the establishment (if not management) of protected areas.

Wholesale landscape change, and observed decline in many species have lent support to the establishment of reserves, not just to protect unique or beautiful places, but as a system to perpetuate the range and diversity of biota and natural systems which may not persist in the absence of such reserves.

For example, in 1967, a scientific committee was established to advise the NSW government on:

> ...the optimum sample of the various ecosystems throughout the State which should be reserved for scientific purposes from those lands still essentially in the natural state, or capable of being restored to something approximating the natural state. (Tom Lewis, quoted in Whitehouse, 1990, p.13)

The resulting analysis documented how well each ecosystem was represented within reserves across NSW (Whitehouse, 1990). Around the same time, a ten year long Australia-wide study, under the auspices of the Australian Academy of Science, identified approximately 900 floristic alliances in Australia, documented how well they were represented within reserves, and made recommendations for a national system of ecological reserves (Specht, 1975).

Only since the mid 1980s, however, have most Australian State nature conservation agencies embraced the goal of representing the wide range of ecosystems within each jurisdiction in a system of protected areas (Thackway, 1997). The concept of a representative reserve system stands in contrast to the earlier (but still influential) approach, of reserving the atypical or unique. The representative approach differs not just because it aims to apply a rigorous methodology to reserve selection. It differs first and foremost because it seeks to place quite different attributes within reserves. Rather than finding that which is unique, remarkable or beautiful, a representative reserve system bases its goals on the generality and variety of a region. It seeks to identify the nature and extent of the elements of variation, and then to find areas which will reserve a subset of those elements, sufficient to represent or sustain them, if those outside reserves are lost. It is a kind of Noah's ark strategy.

Decisions about locating and establishing reserves invoke a complex array of social, economic and cultural factors, which need to be addressed within a process for landuse planning. When worthlessness for other uses, and special uniqueness were the main criteria for reserves, this was easily met on an ad-hoc basis. It lent itself more easily to popular decisions. Selecting nature reserves to represent biodiversity implies different procedures for selecting them, and creates a very different set of expectations for them to fulfil. In some cases, for example, biodiversity conservation may conflict with people's expectations of reserves as *parks* managed for recreation.

Using ecological information to propose a set of areas which is representative of the biodiversity of a region is just one part of a complex decision making process. It is important however, because it seeks to ensure that where a limited area can be allocated to reserves, it will make as large a contribution as possible to conservation of the biota of the region. As would be expected with a relatively new approach, however, there are many scientific and technical obstacles to achieving this goal. This project addresses some of these technical questions, but does not explore any further the equally important social aspects of reserve planning. The next section briefly outlines the problems, and then describes how the current project fits into this context.

1.3 Conservation goals

A systematic approach to reserve selection is important. Ad hoc creation of reserves as has been the norm in the past (Thackway, 1997) may actually get in the way of creating a system of reserves which is representative of the diversity of an area. As Pressey (1994) has shown, bias caused by a preference for selecting particular kinds of areas may lead to a large expansion in the total area reserved, while the variety of unrepresented biodiversity does not decline substantially.

With this in mind, the approach generally used to plan a representative conservation reserve system, is to define conservation goals for those elements of biodiversity which are understood to require protection, and then to systematically identify a set of areas which would meet those goals.

As mentioned above, directly sampling and mapping the full range of known elements comprising biodiversity is difficult, impractical, and arguably impossible at a regional scale. Biological surveys generally cover only a small proportion of the area of a region leading to both geographical and taxonomic gaps in information about the distribution of biodiversity in the region (Ferrier and Watson, 1997). The approach commonly employed to get around these knowledge gaps is to use surrogates for biodiversity. Surrogates, in this sense, are attributes which can be mapped in a continuous coverage of the area of interest, such as climate, lithology, forest type or broader ecological classifications, and which are correlated to patterns in the distribution of biota in the region. They are discussed in detail in the following chapter. Surrogates may be used in reserve selection in the expectation that a representative sample of variation in the surrogate will also encompass the variation within other elements of a region's biodiversity. In this approach conservation goals for a representative reserve system are set with the aim of creating a representative sample of the classes within a surrogate.

Formal targets of this type have now been defined in conservation policies at an international and national level. Internationally, the World Conservation Union (IUCN) has adopted a target that protected areas should cover at least 10% of each biome by the year 2000 (IUCN, 1993). In Australia, the Regional Forest Agreement process requires that reserves in each region assessed include 15% of the pre-European distribution of each forest ecosystem (JANIS, 1997). The National Reserve System program extends this approach to the rest of Australia, by classifying the country into bioregions and providing Commonwealth funding for establishment of reserves to create a representative sample of each bioregion (Thackway and Cresswell, 1995).

The adoption of formal reservation targets such as those in the RFA process completes a transition in the role of reserves, from the kind of far-sighted but ad-hoc philanthropy described above, to a set of complex obligations, to be fulfilled within the tight constraints created by other land uses.

How well the resulting reserves actually include particular elements of that region's biodiversity depends to a significant extent on how well the surrogate used captures the variation within that element. It is thus important to be able to evaluate the effectiveness of the surrogate used against other measures of biodiversity. Despite the central role of surrogates in reserve planning, there has been surprisingly little evaluation of the effectiveness of using particular surrogates as a basis for reserve selection (Ferrier and Watson, 1997).

1.3.1 This project

A wide variety of surrogates have been proposed and trialed for use in reserve selection, but currently the dominant approach used for systematic reserve planning exercises in Australia is based on vegetation classes (JANIS, 1997).

Consider a hypothetical situation, in which the surrogate classes defined for a given region were each internally quite homogeneous, but distinct from each other. If this were the case, then a sample of each class should include the full diversity of the region. We would only need to ask how much of each class would need to be retained for long-term viability.

Reality is of course quite different to this — spatial variation in taxonomic composition is continuous, not grouped into discrete classes or assemblages (Gleason, 1939). Although groups of species are often found to co-occur, the response for each species along an environmental gradient, in terms of occurrence and abundance, is generally independent. This results in a continuum of community composition, rather than a series of discrete classes (Whittaker, 1975). A classification seeks to capture the key elements of this variation, but could never do so perfectly, even with perfect knowledge, because significant transitions will occur at different places, depending on what elements of biodiversity are being considered. The closer a classification is to capturing this variation, however, the more effective it will be as a tool for selecting areas to represent biodiversity in reserves.

This project uses a case study to examine how well a reserve system representative of mapped vegetation classes for a region is representative of its biodiversity. The study is restricted to one element of biodiversity, vascular plants, and to a single region, which takes in much of East Gippsland.

Specifically, the ecological vegetation classes mapped for East Gippsland were used to select reserves which met the internationally accepted IUCN target of 10% representation for each class (IUCN, 1993). The reserves were then evaluated against records from flora quadrat sites across that region. The quality and quantity of the flora survey data collected for East Gippsland provide a good opportunity for reserve evaluation, as Kirkpatrick (1998) noted in his recent review of the RFA process.

A computerised reserve selection algorithm which operated on spatial data for the study region was used to generate a range of reserve options which each met the 10% representation target. Reserve selection algorithms are tools which help decision makers to identify such sets of areas which simultaneously meet a number of conservation goals. Their development and use are discussed in chapter 2.

Going one step further, the project then compares the above approach to selecting a reserve system using the plant species records, bypassing (or complementing) the surrogate approach, to directly choose areas which will maximise the species richness of the reserve system. Given the known weaknesses in using surrogates, can we improve the reserve selection process by using species records directly in choosing areas for reserves?

With a focus on meeting particular representation targets, it is easy to lose track of the importance of the spatial configuration of the resulting reserve system. Large and well connected reserves are important to buffer the effects of surrounding land management, to support viable populations, to provide pathways for dispersal, and room for species assemblages to move and change over time in response to disturbance, succession or climate change. The approach used in this project shows how a given set of representation targets can be achieved within a significantly smaller area, if no restrictions are placed on the size and spatial configuration of reserves. Such a strategy however ignores the impact of spatial configuration on the ecological role of reserves and their effectiveness for long term biodiversity conservation. The comparison of reserve selection techniques in this study thus considers the spatial configuration of the modelled reserves as an important factor alongside their representation of biodiversity.

Some problems with the analysis performed are considered in chapter 5 in order to define the type of conclusions which can be drawn legitimately, and to propose some methodological issues which could be considered in future work.

2 Background and Theory

2.1 Conservation goals

To plan a representative reserve system it is necessary to define what attributes the reserves are to represent, and how much of each attribute is considered to be a representative sample. These attributes and amounts sought for a reserve system are commonly known as conservation goals. Where the starting point is a goal of representing the biological diversity of a region in reserves, the first step is to specify what is meant by biodiversity, and devise methods of mapping and measuring the occurrence of its various elements.

2.1.1 What is biodiversity and how do we measure it?

The concept of *biodiversity*, understood most broadly as *the variety of life*, is used very widely in discussions of nature conservation, in both science and public policy. It is gradually becoming current outside specialist circles, and can help to engender a broad vision of the living world which goes far beyond particular charismatic or commercially useful species. The weakness of using such an all-encompassing, unifying concept however, is that it is very hard to grasp and apply in a practical sense.

Biodiversity is commonly defined by a hierarchy of three levels:

- genetic - the genetic variation of the individual plants, animals and micro-organisms that inhabit the earth. Genetic diversity occurs within and between the populations of organisms that comprise individual species as well as among species;
- species - the variety of species on the earth; and
- ecosystem - the variety of habitats, biotic communities and ecological processes (Heywood, 1994; Commonwealth of Australia, 1996).

Note that in reality the boundaries between these levels are often far from distinct (Gaston, 1996b). They might best be understood as convenient ways to conceptualise a continuum of variation at different scales.

In technical use, the way biodiversity is defined, measured and described depends on the purpose for which it is done, and the scale at which the information is to be used. It could range, for example, from describing genetic variation among seed of a single chenopod species, gathered in an ant nest (Peakall et al., 1993), to comparing species richness within two families of butterflies at a global scale (Vane-Wright et al., 1991), to mapping dominant tree species in a region from aerial photos for timber production purposes.

For this project, the measures of interest are those which can be applied to conservation planning at a regional scale. Measures used for this purpose need to provide spatial information, to enable comparisons between different locations, and would ideally be consistently applied to the whole of the region.

In a theoretical situation of perfect information, one might plan a reserve system based on full knowledge of the distribution of all ecological communities and species found in the region, their patterns of internal variation and of change over time. One would be aware of the centres of abundance for each species, and their genetic variation across the region, their edaphic and biotic habitat requirements, responses to and patterns of disturbance, and patterns of co-occurrence with other species. Ecological communities might even be mapped as patterns of continuous variation, rather than as categories. Information from beyond the region being planned for would also be used, for example, to assess the significance of the region for the range of each taxon, and to identify endemism.

The reality is of course very far from this ideal. On-ground biodiversity survey work is time, labour and vehicle intensive, and thus very costly (Margules and Austin, 1991). For example, some fauna are only readily observable in certain seasons or by certain observation methods (Parris, 1999), while others may be migratory. Many plants can only be reliably identified when in flower (Entwisle et al., 1993). Although often overlooked, the invertebrate fauna is typically more diverse, more locally variable and less well described than the plants or animals (Mummery and Hardy, 1992). Even a relatively small area may thus require multiple visits by experts in different fields to adequately sample its biota. It is thus understandable that in a large, biologically diverse and sparsely populated country like Australia, biological survey data may provide quality information about particular taxa or particular locations, but cannot come close to providing a spatially continuous description of the variation in biota across a region. There will generally be geographic and taxonomic gaps in data from even a well studied region (Ferrier and Watson, 1997).

The approach usually taken, where a spatially continuous description of a region is needed, is to rely on forms of spatial data which can be conveniently used to generate continuous coverage of a region, such as satellite images, aerial photo interpretation, geology, topography and modelled climate surfaces. Correlations between these attributes and site records of particular aspects of biodiversity are used to generate models or surrogates which interpolate the distribution of species, vegetation types or other elements of biological diversity between ground-surveyed locations.

It is arguable that considerably more resources should be put into primary field data collection. Even this however would not remove the need to interpolate between surveyed locations, but could (assuming good survey design) certainly improve our ability to do so reliably.

Given the limitations on field data collection described above, most projects which seek to select areas representative of the biodiversity of a region, are actually selecting to sample variation in a surrogate mapped for the region. It is assumed that this variation would incorporate variation in other elements of the biota. Within a regional biodiversity conservation study, some selected taxa might be considered individually, perhaps including the better studied vertebrate fauna, along with known populations of rare or threatened species (AHC and CALM WA, 1992; RFA Steering Committee 1996). The majority of the taxa in a region would however be considered only indirectly, through variation in the surrogate used.

2.1.2 Surrogates for biodiversity

The choice of surrogate, and its effectiveness at capturing variation in particular elements of biodiversity, is thus a central element in reserve selection. Proposing a reserve system to protect areas representative of the diversity of the natural environment is a widely accepted aim. It has real meaning however, only when one can specify in a measurable form, what it is that the reserves will seek to represent.

> The apparent adequacy of the reserve system and the areas selected for new reserves depend very much on the way in which biophysical diversity is defined. (Pressey, 1990, p.70)

The range of surrogates proposed or used includes land systems, bioclimatic domains, environmental ordination, vegetation mapping, as well as various combinations of these methods.

One way to understand these approaches is in terms of the observable attributes on which they are based. A continuum can be drawn, which extends from classifications relying purely on abiotic parameters, to those using entirely biotic or taxonomic parameters. This categorisation draws on Austin and Margules (1986).

Abiotic ⟵⟶ Biotic

Climate — Geology — Topography — Soils — Vegetation structure — Dominant plant species — All plant species

Most surrogates used in reserve design use attributes from various points on this spectrum. Data for attributes further to the left are generally more readily available at regional scales, while those to the right require increasing amounts of location-specific effort to collect. The approach which is most appropriate for a given region may depend on the scale at which the analysis is to be performed.

> Biological variation can be defined at a range of scales. The larger the area for which context is required, the more general becomes the nature of the target biological unit. For example, at a local scale, the representativeness of species' distributions can be assessed, while at a regional level, applications have examined vegetation formations and communities together with their associated habitats. (Mackey et al., 1989, p.281)

A description of these different approaches follows.

Environmental domains and ordinations

An abiotic environmental classification analyses variation in terms of physical environmental regimes which mediate the growth of plants and other organisms and act as limiting factors for their distributions. Statistical analysis of temporal and spatial patterns in the availability of water, nutrients, radiation and temperature can be used to create classes populated by areas experiencing similar environmental conditions (Mackey et al., 1989), and thus, in general supporting similar biota. These classes, generally known as environmental domains, have been used in regional studies to assess the representativeness of reserve systems (Mackey et al., 1989) and to propose new reserves (Richards et al., 1990; Kirkpatrick and Brown, 1994).

Environmental ordination is another approach which uses variation in the parameters of the physical environment to derive a surrogate for biodiversity. In this approach, parameters measured for the physical environment are assessed in relation to species abundance, to derive vectors in multi-dimensional environmental space which are correlated to abundance of the species in question (Faith and Norris, 1989). This method can be used to interpolate species or community occurrence between sampled points, by first locating each point of interest within the ordination space. In this way, the representativeness of reserve areas can be assessed in terms of their distribution within the ordination space. An advantage of ordination methods over environmental domains and other methods which define classes, is that they provide information about the relative similarity of sites, rather than just whether they are in the same class or not. This helps to overcome problems about how many classes to define, and where to draw the boundaries between them (Faith and Walker, 1996).

Biotic surrogates – vegetation mapping

Biotic surrogates involve the use of data collected about some aspect of the living environment as a surrogate for variation in other aspects of the biota.

In this paper, most attention is given to the biotic end of the spectrum presented above, particularly vegetation mapping. This is not because it is necessarily a better approach, but rather because vegetation based classifications are the surrogate of choice for recent and current reserve planning processes in Australia, and thus warrant some careful evaluation.

Vegetation mapping techniques used in Australia, vary in the extent to which they emphasise structural or floristic attributes of the vegetation. Broad-scale mapping of vegetation structural characteristics (such as height and canopy cover) is available for large areas of Australia, largely collected by aerial photo interpretation (API). Such structural mapping, combined with data on the floristic composition of the canopy, forms the basis of forest type mapping used in planning timber production and other land management in most Australian States (Sun et al., 1996). With considerable State resources put into this form of mapping, there is a good coverage of many regions of Australia, although classifications often show an emphasis towards commercial timber species (Sun et al., 1996).

Moving further toward the biotic end of the continuum, a number of floristic classification systems have been developed based on numerical analysis of flora site data. A typical approach is that used in Victoria (Gullan et al., 1981), while more recently computer-based numerical classification techniques such as PATN (Belbin, 1995), have been used to identify groups of quadrat sites with similar species composition. These groups of sites form the basis of floristic sub-communities. Sub-communities may be aggregated into floristic communities which share a common core of species, but with a lesser degree of floristic homogeneity than a sub-community.

There appears to be a tension however between the relative fidelity of this approach to floristic variation, and problems of practicality for on-ground implementation. Austin and Margules (1986) describe how communities defined from numerical analysis of presence/absence data from the NSW south coast had to be manually reworked to produce useful classifications. The concerns they cite about their original floristic classification include undue influence in classifications given to 'obscure ground layer herbs' (Austin and Margules, 1986, p.59). They described their solution as follows:

> [I]n order to make the classification a practical one for conservationists and foresters it was modified subjectively using quantitative data (basal area) from the forest trees. (Austin and Margules, 1986, p.59)

Austin and Margules' experience suggested that a classification which is based purely on taxonomy may be problematic if it fails to 'ring true' to our perception of vegetation, in which the large and more visible elements dominate, and also because it treats all taxa equally, regardless of the magnitude of their biomass or influence on ecological processes. A solution to this problem, used by Richards et al. (1990) was to perform separate community classifications for canopy and understorey species.

Some of the differences between these two approaches discussed above, forest type and floristic community mapping, are summarised in Table 1.

Table 1 - Differences between forest type and floristic community classifications.

Forest types (from API)	**Floristic communities (from survey quadrats)**
Emphasise dominant canopy species Ignore most understorey variation	Treat all species equally Reflect finer scale understorey variation which may not be seen from the air
Continuous coverage	Classifies surveyed sites only, and thus may require use of API to extend to mapped polygons
Easier to understand and recognise in the field	Harder to recognise, but may draw attention to important floristic distinctions

2.1.3 Ecological Vegetation Classification

The general approach

In recent years, State agencies have developed approaches which seek to combine the strengths of the floristic community, forest type and abiotic classifications described above, to produce a general vegetation classification system for conservation use. Examples of this approach include *Ecological Vegetation Classes* in Victoria (Woodgate et al., 1994) and *Forest Ecosystems* in New South Wales (Keith and Bedward, 1998).

These and similar vegetation mapping systems developed around the country have formed the primary surrogate for reserve selection used in the Regional Forest Agreement process. As a result, they represent the current preferred surrogate in Australia, which is being used and implemented in real life reserve design processes.

Classification begins with pattern analysis of floristic similarity at quadrat sites to define floristic communities, as described above, which are effectively groupings of quadrat sites. To interpolate from points to mapped polygons defining the boundaries of each class, a variety of spatial modelling techniques are used, which associate the floristic communities to spatially continuous data. This data includes API-based forest types, vegetation structure, parent material and a range of climatic, and topographic parameters. Ecological vegetation classification thus draws on the full range of attributes in the biotic-abiotic continuum presented above.

The classes produced have a floristic component, but also reflect vegetation structure and dominant canopy species, which Austin and Margules (1986) considered important for practical recognition and application. This approach could be said to take the best elements from of the two approaches contrasted in Table 1 above, forest type mapping and floristic communities.

The complexity of linking the floristic and forest type information, requires a large element of subjective expert judgement, and as discussed in chapter 4, this may cause problems for the transparency of the resulting classification.

2.1.4 How well do surrogates work?

Despite their widespread use, surprisingly little research has been done on evaluating the effectiveness of surrogates for biodiversity. A number of comparisons between modelled and actual distributions of individual species have been published (for example Williams, (1991) and methodologies for this are well developed, but this is not the case for surrogates which seek to represent a broader range of taxa.

One study (Kirkpatrick and Brown, 1994) examined the use of environmental domains for reserve selection by designing potential reserve systems for Tasmania to create a representative sample of either environmental domains, or of a range of communities and significant species. The areas selected by these two approaches were found to overlap to a significant extent. They concluded however, that "reservations selected on the basis of environmental domains are likely to represent the more widespread biotic attributes in a reasonably satisfactory manner, but they will not capture many of the rarest species and communities." (Kirkpatrick and Brown 1994, p.222).

This conclusion should be interpreted not so much as a criticism of environmental domains, but rather to highlight a need to pay special attention to known locations of rare entities, whose distributions may have a large stochastic component or be limited by factors other than those embodied in the surrogate (Faith and Norris, 1989).

Ferrier and Watson (1997) developed a methodology for surrogate evaluation, and used this to test a wide range of biotic and abiotic surrogates against site records from forested north-east NSW, covering a range of taxonomic groups. Sites selected to represent the diversity in a given surrogate, were evaluated by analysis of the number of species accumulated at those sites. A better surrogate would require less sites to reach a given level of species richness. Their broad conclusions were that:

- all surrogates tested performed poorly for ground-dwelling invertebrates;
- the poorest performing surrogates were those derived purely on abiotic environmental data;
- forest type mapping generally outperformed other types of environmental classification and ordination for both vertebrate fauna and vascular plants;
- the best performing surrogates were those which used spatial models of distribution of taxa in one group (eg canopy trees) as a surrogate for other groups (eg reptiles, understorey flora).

Ferrier and Watson (1997) did not suggest that any of the surrogates was sufficiently effective to be used alone as a basis for reserve selection. Further, they warn that these results should be applied cautiously to other regions where there may be significant differences in both ecological relationships and also in the quality and quantity of environmental and biological data.

While the evaluation of surrogates is at a relatively early stage, their use in reserve selection is progressing apace. Vegetation classes of the types described in section 2.1.3 were not evaluated by Ferrier and Watson, but are central to reserve planning in the RFA process. This is why it is important to know how well we are representing the taxonomic diversity of a region, when we reserve a representative sample of vegetation classes.

2.2 Reserve design (spatial configuration)

So far the discussion in this chapter has considered ways of defining biological diversity and mapping its variation in space, so that a representative sample may be identified. If the purpose of a reserve system is to support viable populations of the biota of a region, it is not sufficient to simply have a set of areas with the required range of biological attributes. The location, size, shape and connectedness of reserve areas must also be considered, in order to create landscapes which not only sample the diversity of a region, but can also sustain it.

The greatest threat to biodiversity worldwide is the human induced alteration and removal of habitat (Noss and Csuti, 1994). This process is characterised by two related processes: reduction in the area of habitat, and increasing fragmentation and isolation of those areas which remain.

2.2.1 The effect of fragmentation on populations

The fragmentation of habitat has several effects on the populations living within that habitat. Firstly, less habitat area means that less resources will be available, and thus total population sizes will decline. Secondly, smaller individual habitat areas support smaller complements of indigenous species. Where the size of a habitat area is reduced, a process of species relaxation occurs whereby some species living within the remaining area become locally extinct over time, as species richness ‘relaxes’ to a reduced level appropriate to the size and available resources of the remnant (Saunders et al, 1991). This process may take some time, as adults may survive even if the remnant cannot support the full reproductive life cycle. Species diversity will not necessarily decline, but original species in the area may be progressively replaced by other taxa, often exotic, which are more suited to the new conditions (Bennett, 1990).

Although habitat suitable for a given species is often naturally patchy, the spatial isolation of habitat areas is greatly increased by human induced habitat fragmentation. Small populations are more vulnerable to local extinction, due to factors such as inbreeding depression and demographic stochasticity (Soulé and Simberloff, 1986). The distance and degree of connectivity between such patches may be crucial to whether populations function as isolated relicts, or as elements of a broader (and hence more viable) metapopulation. The persistence of a population thus relies on habitat areas which are either sufficiently large to support a viable population, or sufficiently close or connected to facilitate genetic interchange and recolonisation to reverse local extinctions.

Proximity however, does have some disadvantages. The closer two sites are together, the more similar their physical and biological properties are likely to be (Kunin, 1997), and the more likely they are to affected by the same events including spatially correlated disturbances such as disease and fire (McCarthy and Lindenmayer, 1999). Including more widely separated sites within a reserve system can thus help to increase the ecological or taxonomic diversity of the reserve system, and also provide some insurance in the event of destruction of a local population.

Population viability analysis (PVA) is a technique used in conservation planning to estimate the likelihood of persistence of a population under particular conditions, and thus to identify the conditions most conducive to its survival (Schaffer, 1990; Possingham et al., 1993; Brook et al. 2000). A number of studies have applied PVA to particular plant species (for example Menges, 1990; Burgman and Lamont, 1992; Nantel et al, 1996) to draw conclusions about the conditions likely to result in persistence of a species in question. While this approach may be successful for a particular, well studied species, it is less easily applied where the aim is to sustain a broad range of taxa within a given area. Even where the life history and requirements of species are understood, the response to influences such as disturbance and fragmentation varies widely between species depending on factors such as their reproductive and dispersal characteristics, use of the matrix, and the spatial scale at which they interact with the environment. A landscape which is seriously fragmented for one species, may cause few problems for another.

Given a lack of detailed life history knowledge for many species, where the reserve system is intended to cater to a broad range rather than a particular species, it is necessary to fall back on some broad principles for reserve design.

At the most general level, principles for reserve configuration for the maintenance of viable populations would include the need for large, diverse, well connected reserves, but also for well separated replicates which encompass environmental and genetic variation. Separated locations provide some insurance in the event of elimination of populations, for example by fire or disease.

2.2.2 Edge effects

The reduced size of patches in a fragmented landscape is compounded by the increased proportion of habitat area subject to edge effects. The boundary between areas of retained vegetation managed for conservation and surrounding areas used for other purposes may appear as a sharp line on a map or satellite image. In terms of its effect on habitat conditions however, the edge may be seen as a broad transition zone between the conditions of the matrix and those of the remnant.

The microclimate near the vegetation edge is affected by the conditions in the adjoining matrix in several ways (Saunders et al., 1991). Firstly, clearing changes the energy balance of adjoining areas, causing changes in the radiation and temperature regimes. In a forest, shade tolerant species may become restricted to interior areas of the remnant. Secondly, wind patterns will reflect those prevalent in the adjoining vegetation for a significant distance into the remnant. Thirdly, depending on position in the catchment, changed landuse can have a great effect on surface and ground water flows, and on the transport of sediment and nutrients.

Edges may also facilitate different interactions between species (Gardner, 1998; Brand and George 2000) and provide an opportunity for the invasion of exotic species, for example from farmland or roadsides (Pivello et al. 1999). While some species are more prevalent near edges, other species are likely to be found only in the interior of a vegetation remnant, beyond the range of edge effects, and these species may be most at risk from habitat fragmentation (Canaday 1997; Luck et al. 1999). For example, one study found that the size of the area not subject to edge effects within a remnant, accounted for the majority of variation in native species richness (Dunstan and Fox, 1991).

These edge effects mean that many species require reserves with interior areas, well buffered from edges. The shape of a reserve will help to determine how much of its area provides such habitat. For example, a narrow, elongated reserve would contain far less interior than a circular reserve of the same area. The proportion of a reserve subject to edge-effects can be described or calculated using the 'perimeter-area ratio'. A high perimeter-area ratio indicates that the reserve is likely to be highly influenced by the conditions of surrounding areas (Meffe and Carroll, 1994).

2.2.3 Balancing cohesiveness and diversity

The factors described above point to a need for large, well connected reserve areas with low perimeter–area ratios in order to maximise the viability of populations and minimise the impact of adjoining land management on the ecology of protected areas.

The goal of maximising the diversity of ecosystems and species represented in the reserve, however suggests a different approach to reserve design. A number of smaller, reserve areas spread widely across the geographic and environmental space of the region would generally be able to include a greater range of species and habitats within the same total area. Protecting several separate populations of a vulnerable species may also provide insurance against the possibility of a single fire, storm or epidemic causing an extinction.

The relative merits of these two approaches were considered at length in what became known as the *single large or several small* (SLOSS) debate (Meffe and Carroll, 1994). The SLOSS debate eventually concluded with a recognition that both a minimum viable reserve size and the distribution of reserve areas across a region are necessary (Soulé and Simberloff, 1986), and that the best approach will depend on factors specific to each case (Higgs, 1981). Although the debate over these principles is largely in the past, the trade-off between cohesiveness and diversity remains a crucial factor in selection of reserve areas.

The case study described in this paper uses a new approach to explore this trade-off between cohesiveness and representativeness in reserve selection.

Several authors have pointed out that such questions of spatial configuration are largely theoretical, because scientists and land managers are generally faced with landscapes which are already highly fragmented (Saunders et al., 1991), or selected for protection based on factors other than ecology (Soulé and Simberloff, 1986; Pressey, 1990). I would argue however that in Australia there are sufficient areas in a relatively natural state, where there is potential to apply ecology to the planned management of landscapes, to justify the relevance of questions of spatial reserve design[1].

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[1] For example, large scale land use planning exercises such as RFAs and the Cape York Peninsula Land Use Study have recently presented opportunities to apply (or at least present) ecological principles in the spatial design of protected areas.

2.3 Systematic approaches to reserve selection

The selection of areas which meet conservation goals, maintain good spatial reserve design and consider the impact on other potential land uses, is a very complex task. For this reason, a number of systematic techniques have been developed over the last two decades to select or prioritise areas for reservation. Although approaches have varied, the aims in each case are: (i) to identify areas which will best represent the environmental values of the region in question; and (ii) to do so in a way which is consistent, empirically based, and can be used to justify the resulting reserve proposals.

These techniques, known as reserve selection algorithms, have developed from simple rating systems into relatively sophisticated computerised optimisation programs. As these techniques have developed, problems or factors which must be taken into account for good reserve selection have been successively identified and addressed in subsequent selection techniques.

Representation, cost and spatial configuration are factors which can be used to define the desired reserve system. Other factors such as efficiency, complementarity, irreplaceability and optimality describe important considerations in the selection process. Each of these terms is explained below as it enters the story of selection algorithm development. Reserve selection algorithms address a related but distinct class of problems from those concerned with setting conservation goals. Questions in conservation biology related to species' habitat requirements, threatening processes, sizes and shapes of protected areas, and population viability analyses, are inputs which are used to create the information and define the conservation goals used in reserve selection algorithms.

The algorithms themselves, although designed for an ecological question, are essentially value neutral. Their task is to select a set of areas containing target amounts of particular mapped values, within specified conditions. They are in general applicable regardless of whether the values in question are based on land systems (Pressey and Nicholls, 1989a) environmental domains (Mackey et al., 1989; Bedward et al., 1992), sampled species locations (Margules et al., 1988), modelled species distributions (Margules and Stein, 1989), or maps of suitable habitat. They could even be used to select sets of areas with attributes required for urban or agricultural use.

2.3.1 Efficiency

The concept of efficiency is important for the comparison of different reserve selection techniques. Efficiency provides a tool to gauge the ability of different selection methods to choose sites or areas which sample the diversity of a region while reserving a limited area or number of sites. It indicates the proportion of area or sites in a region which must be selected using a given selection method, to include the required conservation values within reserves. A simple index of sampling efficiency is defined by the formula:

$$E = 1 - (X/T)$$

where E is efficiency, X is the number of sites, or total area needed to contain a required amount of specified attributes, and T is the total area or number of sites available for selection (Pressey and Nicholls, 1989b). The resulting index ranges from 0 to 1, where 1 indicates highest efficiency, and 0 indicates a reserve which requires all of the area available in order to achieve conservation goals. This efficiency index enables direct comparison between selection methods applied to the same area, targets and data. Comparison between different areas is complicated by the fact that the particular characteristics of the conservation values, the selection units and the reservation targets will help to determine the proportion required to meet conservation targets.

It is important to note that efficiency does not imply adequacy. That is, efficiency is no guarantee that the resulting reserve system will be adequate for species and ecological processes to persist in the long term. Adequacy requires setting appropriate conservation goals, based on good ecological understanding, and the willingness among decision makers to implement such a system. As Pressey et al. (1994, p.243) put it "[i]n most regions, only a small proportion of the land will ever be dedicated to nature conservation so efficiency can determine the likelihood of achieving a reservation goal in the face of limited resources and competition with alternative land uses".

2.3.2 Complementarity

Scoring the conservation value of individual sites – a non-complementary approach

A simple approach to reserve selection is to assess the relative conservation value of areas within a region of interest, giving each site a numeric rating or score (Rabe and Savage, 1979), and then recommend that the highest rated sites be set aside for conservation.

For example, Goldsmith (1975) describes a system to score the conservation value of remnant areas by assessing their value in terms of extent, rarity of community type, plant species richness and animal species richness, while Gehlbach (1975) also considers the successional stage, level of human impact and educational value. Each such system includes a method for rating and weighting the various conservation values and combining them into a single score for each site (see Figure 1). Using such a rating system, the places with the highest conservation value can be given priority for reservation, with those progressively lower on the list being added depending on the amount of land which can be allocated to conservation. Scoring systems of this type can contribute consistency and objectivity to the assessment of conservation values and provide a clear justification for conservation of particular areas within a land use planning process.

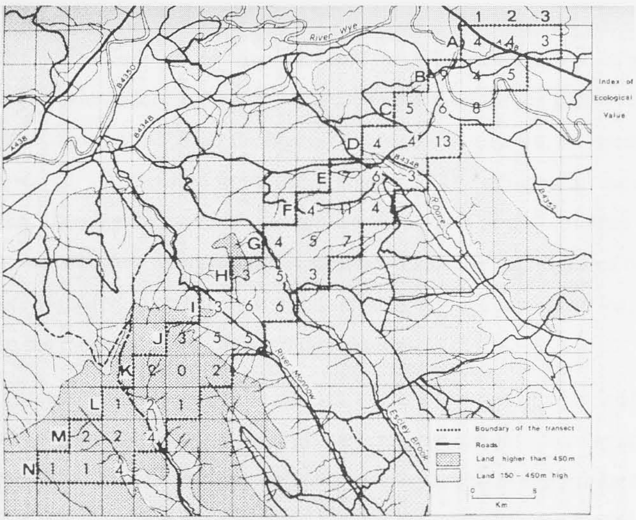


Fig. 1. Location of the transect between the Black Mountains in Breconshire and the Wye Valley, Herefordshire, showing the 42 grid squares used for recording.

Figure 1: A map produced using a multi-criterion site scoring approach in which each area is given an independent score (Goldsmith, 1975, p.93). This approach is objective and repeatable, but its weakness is that by treating each site independently, it cannot identify sites which complement each other to represent the range of environmental values identified for the region.

The need for complementarity

When a reserve system is being designed or evaluated for biodiversity conservation, it is not primarily the value of each individual area which is important, but its contribution to the reserve system as a whole in terms of its protection of the biodiversity and other significant attributes of the region. For this reason, methods as described above, which rate each area independently for their conservation values may prove inefficient at selecting the set of areas which adds most to the conservation estate (Kirkpatrick, J., 1983). A given area might be placed at the top of the priority list because the species or communities which it contains are endemic to the region and poorly reserved, but as Kirkpatrick (1983) illustrated for an area of eastern Tasmania, the areas second or third on the priority list might duplicate the species in the first area, while others are missed altogether. What is needed is a system which selects areas which complement rather than duplicate the values in areas already reserved. In the context of reserve selection, complementarity refers to the selection of areas which include previously unsampled (or undersampled) features to the reserve system. A complementary selection increases representativeness of the reserve system.

Iterative selection

Kirkpatrick's (1983) solution is iterative selection. In this method, as well as giving scores to the attributes valued for conservation, targets are set for the appropriate level of reservation of each attribute. After the highest rating area is selected, it is treated as nominally reserved, and removed from further selection. The rating of all the other areas is recalculated, allowing for the improved conservation status of the attributes already reserved, and the area now rating highest is selected. Once the required area or number of occurrences for a given attribute have been selected, that attribute is no longer considered in the rating of areas still awaiting selection.

By treating conservation value not as a constant, but as a function of what is already protected, iterative methods provide more efficient reserve options. Similar approaches have been used, for example, to select wetlands to reserve species and habitat types (Margules et al., 1988) and to identify world regions of highest priority for the conservation of swallowtail and milkweed butterflies (Vane-Wright et al., 1991). In a comparison of individual site scoring methods with iterative methods, the iterative methods were more efficient than non-iterative procedures (Pressey and Nicholls, 1989b).

A major theme in research on iterative reserve selection techniques since Kirkpatrick (1983) has been experimentation with different heuristics. An heuristic, in this context, is a rule to guide the order in which areas are selected. For example, the wetland study mentioned above (Margules et al., 1988) used the following steps:

1. Select all wetlands with any species which occur only once.
2. Starting with the rarest unrepresented species, select from all wetlands on which it occurs, the wetland contributing the maximum number of additional unrepresented species.
3. Where two wetlands contain an equal number of unrepresented species, select the wetland with the least frequent group of species.
4. Where two or more wetlands contribute an equal number of infrequent species, select the first wetland encountered.

Pressey et al. (1997) compared a broad range of heuristics which used different sequences of rules by which to select the next area from those available. These included:

- rarity of the attributes of the site;
- richness – the number of attributes of value on the site;
- richness per unit area;
- maximum rarity – the rarity of the rarest attribute at the site;
- average rarity – the average rarity of all attributes at the site.

They concluded that the although some sequences of rules were more efficient than others, the best rules in a given situation varied depending on the data and conservation goals used.

2.3.3 Irreplaceability

There will often be considerable flexibility in meeting reservation goals by selecting different sets of areas to include the required attributes. Recognising this, Pressey et al. (1994) proposed irreplaceability as a measure of the extent to which particular areas are essential to a representative reserve system. One way of defining irreplaceability of a given site is by "the extent to which the options for a representative reserve system are lost if that site is lost" (Pressey et al., 1994, p.243). Irreplaceability is a dynamic measure, which depends on the specific conservation goals used. The higher the degree of conservation sought in a region, the more sites would be essential in achieving that goal. One index of irreplaceability of a site, is the proportion of all combinations of sites which meet the requirements of the reserve design process, in which that site is included. Except for very simple reserve selection problems it is not possible to run through all permutations of areas selected, to calculate the index in this way, but other estimates of irreplaceability are possible (Pressey et al., 1994).

2.3.4 Optimality – Linear programming

An alternative to iterative analysis, is to formulate the reserve selection problem as an optimisation algorithm using linear programming techniques (Cocks and Baird, 1989; Underhill, 1994; Church et al., 1996). The problem definition used by Cocks and Baird is similar to the iterative analyses described above, in that it considers the conservation value of a system of reserves rather than of individual areas. Unlike iterative methods, however, it dispenses with heuristics which guide the order and priority for selection, and instead defines the characteristics of the desired reserve system mathematically. The problem is defined as a function which expresses the extent to which the system deviates from the specified reserve design targets (Cocks and Baird, 1989):

$$Minimise\ D = \sum_{i=1}^{I}(P(i)y(i) + P'(i)y')i))$$

where

D = a measure of the extent to which the reserve system deviates from the overall representation goal

i = each of the reserve targets (i = 1,2...I)

P(i), P'(i) = the penalty for overshooting, undershooting the ith system goal

y(i), y'(i) = the number of units by which the ith goal has been overshot, undershot.

By definition the function has an optimal solution. This is the set of areas which best meets the representation goal, and for which D is lowest, and this will be the reserve design generated using this approach.

The advantage of using a linear programming formula to define the desired solution, is that if a solution is found, it is guaranteed to be the most efficient set of sites that meet all constraints (Underhill, 1994; Church et al., 1996). Heuristic algorithms can make a good decision at each step, but cannot guarantee to find the optimal choice of areas. In comparisons, heuristic algorithms required 5-10% more area than the optimal solutions calculated with linear programming (Pressey et al., 1997).

Although linear programming can provide an optimal solution for a given problem definition, it has so far been far less used and developed in reserve selection than stepwise heuristic algorithms. This is because the computation required is orders of magnitude greater; too great to be feasible for typically complex problems in reserve selection (Kirkpatrick, J., 1983; Pressey et al., 1997; Ball et al., in press). Further, required processing increases exponentially as more targets or constraints are added. Pressey et al. (1996b) argue that strict mathematical optimality is not highly important for most real-world reserve selection applications.

2.3.5 Cost

As well as the benefits of assigning land for conservation, it is important to recognise that there are both actual costs and opportunity costs associated with reserving an area.

Cost will always be addressed in one way or another when real-world decisions are made about establishing new reserves. Including cost up-front in the analysis may reduce unnecessary competition with alternative land uses, and thus increase the likelihood of the resulting reserve proposal actually being implemented.

> To make reserve proposals defensible in the light of competing land uses... the selection of areas for reserves should be as efficient as possible, i.e. the cost of the reserve network, in terms of resources no longer available for other land uses, is not greater than was necessary to accomplish conservation goals.
> (Bedward et al., 1992, p.117)

Depending on the analysis, cost could be used in various ways. It could indicate the potential economic value of extractive uses (such as logging, mining) which would be forgone if the area were reserved, the cost of purchasing private land, or the ongoing cost of managing land once it has been designated for conservation. The Bureau of Resource Sciences has developed cost surfaces for some RFA regions which can be used to incorporate the value of timber and other resources into reserve planning exercises. By including the cost of each site in the algorithm, the algorithm can seek the cheapest solution, balance cost against conservation benefits, or work within an overall cost threshold. Bedward et al. (1992) include a cost reduction phase in their algorithm following initial reserve selection.

2.3.6 Spatial configuration

As discussed in section 2.2, the size, shape and connectivity of reserve areas are crucial to their ability to sustain viable populations. With a few exceptions however, the reserve selection algorithms discussed above or described in the literature select areas containing the required attributes, but without any attention to the location of areas selected, or the resulting spatial configuration of the reserve system.

This may be appropriate where the units of selection are widely separated remnants (Saetersdal et al., 1993), discrete ecological units such as wetlands (Margules et al., 1988) or of sufficient size to be sustainable in the long term (Kirkpatrick and Brown, 1994). Where large areas remain in a relatively intact state, as is the case in some parts of Australia, it is important that the reserve selection process does not needlessly contribute to fragmentation. Some dispersion of reserve areas may be needed to sample the environmental or taxonomic diversity of a region, creating a possible tension between cohesion and representativeness in reserve design (see section 2.2.3). If location of reserves is not addressed in selection however, the degree of fragmentation may be much greater than is required (Nicholls and Margules, 1993).

Bedward et al. (1992) address the spatial design of reserves by allowing the user to manually improve on the result generated by a non-spatial selection heuristic. The user adds or removes areas, to link reserves or improve their shape, and receives information on the effect this would have on cost and achievement of conservation goals. Nicholls and Margules (1993) set their algorithm to resolve a choice between suitable sites by choosing the site nearest in space to a site already selected, while Lombard et al. (1997) choose areas which adjoin previously selected areas over those which don't, if both contain attributes of equivalent value.

2.3.7 Other optimising algorithms

Optimising algorithms are a class of computerised search techniques designed to solve complex problems involving many variables or constraints. They are typically applied to complex staff and resource scheduling problems, such as those faced by airlines. Two types of optimising algorithms which have recently been applied to the reserve selection problem are genetic algorithms and simulated annealing.

Both use an objective function to define the goals and constraints for the reserve system, such as the amounts of each attribute to be represented, the cost and spatial configuration of the system. This function expresses how far a given reserve system is from meeting a set of specified goals. The algorithm seeks solutions which minimise the value of this function

Genetic algorithms are based on an analogy with genetic evolution. A population of potential solutions (sets of reserved areas) is created, and allowed to evolve (Ball et al., in press). Each solution is evaluated against the objective function, and the best solutions are retained, or reproduced with mutations in the next generation. Iterations continue until the population converges on an optimum solution (Beasley and Chu, 1996).

Simulated annealing is a method based on an analogy with the cooling of metals (Kirkpatrick, S. et al., 1983). Applied to reserve selection, it selects reserve areas at random to add or delete from the reserve system. Each random change is checked for its effect on the objective function, to determine which changes will be accepted (Ball et al., in press). It starts off accepting any change to the system, but over time becomes progressively more strict about which changes to accept, rejecting those which increase the objective function by too great an amount. Towards the end of a simulated annealing run, only those changes which directly improve the system are accepted.

Simulated annealing is related to both the iterative heuristic and linear programming approaches described above. Like linear programming it uses a function rather than a heuristic to define the goal being sought, and to express how far away a given reserve configuration is from that goal. It is similar to iterative heuristic selection algorithms however in that it overcomes the computational complexity of calculating an optimal reserve design, by iteratively adding or removing one area at a time to improve the reserve system. Like iterative heuristic techniques, it cannot guarantee an optimal solution.

The stochastic element enables the algorithm to search a much larger portion of the solution space for any particular problem definition than a heuristic algorithm, and thus often find better solutions, but also to solve more complex problems than is currently feasible using linear programming (Ball et al., in press). It also means that rather than generating a single solution, repeated runs will produce different solutions which may find the required attributes in different places where this is feasible.

In comparisons between methods using the same dataset, simulated annealing was more efficient than other iterative selection methods (Ball et al., in press). It came close to or equalled the optimal solution found using linear programming (Csuti et al., 1997; Ball et al., in press).

The major advantage of simulated annealing is its ability to handle very complex problems in a relatively short processing time, such as those incorporating representation goals for many attributes and objectives for the spatial configuration and cost of potential reserve systems (Ball, 1998). A reserve design tool called Spexan which incorporates these functions was used in the current project. Spexan and its simulated annealing approach are described in more detail in chapter 4 of this paper.

2.4 Summary of background and theory

The creation of representative reserve systems which encompass the biological diversity of a region has the potential to be a major factor in slowing the decline of biological diversity. Systematic selection techniques can help to identify sets of areas which will contribute to this aim.

Where reserve selection is based on goals for representation of surrogates for biodiversity, how well the biological diversity of the region is represented in reserves depends on the effectiveness of the surrogate. It also depends on the amount of land which can be selected for reserves.

Although vegetation classes have been widely used in actual reserve selection processes around Australia, little evaluation has taken place, so there is little hard evidence about how well they perform in this role.

Finer scale measures of biodiversity such as flora site records are more difficult to collect, but where they exist, provide an opportunity to evaluate the effectiveness of the surrogates and conservation goals used, or to consider alternative ways of capturing the biodiversity of the region. These two objectives are tackled in the case study described in chapter 4.

In comparing different strategies for sampling biodiversity, it is important to be aware that different selection strategies or representations of biodiversity also imply different spatial arrangement of reserve areas. An effective reserve selection strategy needs to select areas which not only represent the diversity of the region, but also produces reserves which are likely to be manageable and ecologically viable.

3 Aims and scope of the reserve modelling exercise

A regional case study was performed, in which potential reserve areas were selected to meet a set of simplified criteria for area, representativeness and spatial cohesiveness.

3.1 Aims

The aims of the study were:

- to assess the extent to which a reserve system built to be representative of the vegetation classes mapped for a region, is also representative of the known floristic diversity of the region;
- to compare reserve selection based on vegetation classes to direct selection based on flora quadrat records; and
- through the preceding steps, to demonstrate the operation of a new reserve selection tool.

3.2 Scope

The current study applies a highly simplified model of reserve design at a regional scale. It was conducted as a desktop project using data supplied by the Victorian Department of Natural Resources and Environment (DNRE). It is not intended to recommend actual areas which should be protected, but rather to consider the implications of choices of data source and selection method for the cost, shape and representativeness of the resulting reserve system.

The study addresses three factors:

- biodiversity conservation goals which are based on different representations of biodiversity;
- spatial configuration of the resulting reserve systems, using a simple index of reserve cohesiveness;
- the cost of the resulting reserve systems, as a function of area;

3.2.1 Biodiversity conservation goals

Potential reserve systems were selected and evaluated with regard to two representations of biological diversity:

- mapped ecological vegetation classes, and
- vascular plant taxa recorded at survey quadrats.

Other elements of biodiversity often considered in reserve planning processes but not addressed in this study include:

- distributions and habitat requirements of fauna;
- disturbance history and vegetation successional stage (such as old growth);
- presence of introduced species.

3.2.2 Spatial configuration

The model uses boundary length of the reserve system as a simple indicator of adjacency. By seeking to minimise boundary length, preference is given to compact and connected reserve designs with low perimeter-area ratios where possible, given other constraints.

3.2.3 Cost

In this model, area is used as a surrogate for cost. This means that whereas the model seeks to minimise or in some cases cap the cost of reserves, all areas are treated as having an equal cost per hectare.

3.2.4 Factors not addressed in the study

Land tenure

The study is tenure-neutral. This means that no direct consideration was given to whether areas are currently private land, state forest, or national park. The presence of private land did have a limited influence on the results, because the vegetation classification system included a class for cleared private land. Areas in this class were not sought to meet any target, however no restriction prevented these areas occasionally being selected to improve reserve shape or cohesion.

Land use and condition

Real factors which make many areas valuable or unsuitable for reservation, such as current, past or planned land uses, and land condition were also not considered.

Reserve management

The taxa and communities included in reserves will have various management needs in terms of disturbance regimes, responses to invasive species and other threatening processes. These issues are beyond the scope of this project.

Reserve boundaries

The automated selection method used in this and similar studies provides a useful indication of the locations and challenges involved in achieving particular conservation goals. The method used would not be appropriate to define actual reserve boundaries. The planning processes by which real-world reserves are located and their boundaries defined, would need to consider a range of other technical and social factors beyond the scope of this study.

3.2.5 Choice of study area

A region was sought for this study, based on two main criteria. Firstly, it should include a good range of environmental conditions, with a consequent spatial differentiation of plant taxa and associations occupying different ranges. Secondly, good quality biological field survey data should be available for the region. A minor criterion was the author's familiarity with, or ability to visit the study area. Areas considered included Cape York, South-East Queensland, North-East NSW, South-East NSW and East Gippsland.

A region of East Gippsland was chosen which fitted all three criteria well. It has a broad altitudinal range, rising from the coast to over 1200m, has a corresponding broad variation in temperature and rainfall, and includes a good range of topographic features.

Good quality spatial biological data for the region were available from the Victorian Department of Natural Resources and Environment. These included the ecological vegetation classes mapped for the region, and site data from a flora quadrat survey of the region. The flora quadrat dataset in particular is very extensive, covering over 7000 sites, and subject to careful post collection checking against existing data to identify possible errors in the location and taxa recorded for each site.

The density of quadrat sampling is variable however, and in choosing the study boundaries, the least sampled westerly portion of East Gippsland was excluded.

There is no ecological reason to use the state border as the boundary of the study region. It was originally intended to include a small area of south-east New South Wales adjacent to East Gippsland. The NSW area was excluded because the complexity of resolving the different vegetation classifications used in the two states was beyond the scope and timeframe of this project.

The study area is shown on Map 1.

3.2.6 Summary of the scope

This study has simplified the reserve selection problem in order to focus on the main research aims, which relate to the use of different representations of floristic diversity in reserve selection. Excluding factors such as land condition, and restricting the range of taxa considered, removes detail which would unnecessarily complicate the comparisons being made. Essentially the study looks at the region as a series of complex overlapping patterns in the distribution of vegetation classes and plant species, enabling the questions of surrogate choice and reserve configuration to be examined despite the absence of other real-world constraints.

4 Methods

4.1 East Gippsland study area

4.1.1 Biophysical description

The study region covers 754 000 hectares of East Gippsland, bounded to the south and south-east by ocean (Bass Strait and Tasman Sea). It extends from 148°24'E, just west of Orbost, eastwards to Cape Howe, and from 37° 8'S in the north, south to Bass Strait, ranging from sub-alpine mountains, down to extensive coastal flats and tidal inlets.

The region includes all or part of the catchments of the Snowy, Brodribb, Bemm, Cann, Delegate, Thurra, Genoa and Wallagaraugh rivers, all of which drain into Bass Strait.

Elevation ranges from sea level, up to 1291m at Mt Ellery. The Errinundra Plateau at 900-1100m is the most southerly extension of the Monaro plains.

Annual rainfall ranges from a minimum of 750mm in the north-west of the study region, to a very wet 1740 mm on the Errinundra Plateau, with this wide variation driven by the altitudinal range, and by rainshadow effects to the west and north-west of the Errinundra Plateau. The majority of the region averages 900-1050mm annually.

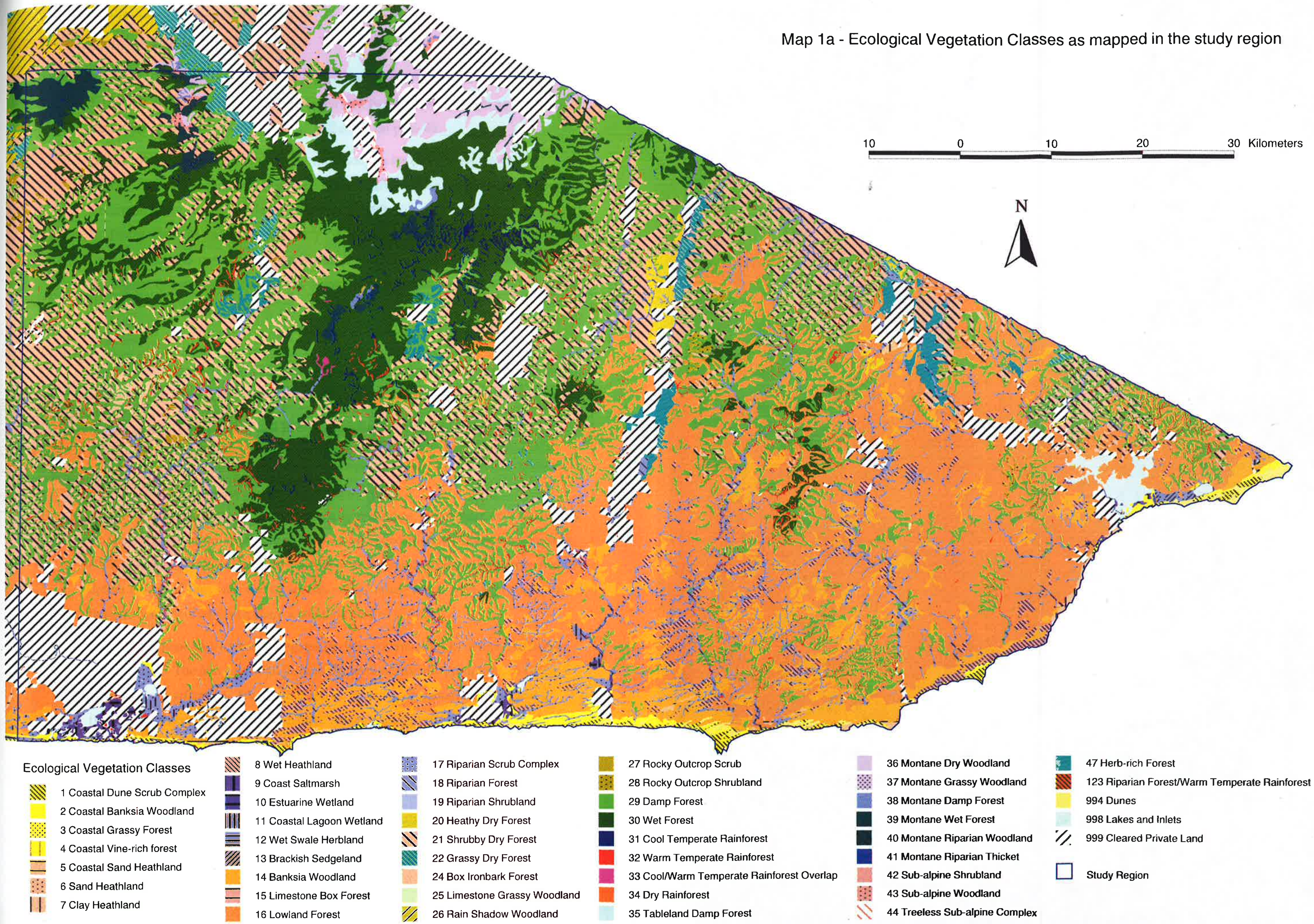
4.1.2 Land Tenure

Over 91% of the study region is in public hands, as State Forest (61%) or National Park (30%). The small amounts of cleared private land in the region are occupied by the towns of Orbost, Cann River and Mallacoota, and agricultural land, predominantly in the Snowy Valley near Orbost, the Cann Valley, and the Delegate and Deddick valleys near the NSW border.

4.2 Description and analysis of the datasets used

The data used for this project were made available by the Victorian Department of Natural Resources and Environment (DNRE).


East Gippsland
Study Region
NEW SOUTH WALES
VICTORIA
SNOWY RIVER
NATIONAL
PARK
NATIONAL
PARK
Errinundra
NP
Coopracambra NP
Croajingalong
NP
Croajingalong NP
Alfred
NP
Lind NP
Nalbaugh
NP
Nungatta NP
Mount
Imlay
NP
Ben Boyd NP
Ben Boyd
National Park
ORBOST
Newmerella
Marlo
Cabbage Tree Creek
Bemm River
Club Terrace
Cann River
Noorinbee
Genoa
MALLACOOTA
Bendoc
Buldah
Combienbar
Delegate
Craigie
BOMBALA
Bibbenluke
Cathcart
Corrowong
Wyndham
MERIMBULA
PAMBULA
PAMBULA BEACH
Lochiel
Yowaka
Towamba
EDEN
Wolumla
Gelantipy
Murrindal
Buchan
Buchan South
Wairewa
Waygara
Lake Tyers
Aboriginal Community
LAKES ENTRANCE
Cape Conran
Pearl Point
Cape Everard
Petrel Point
Rame Head
The Skerries
Sandpatch Point
Little Rame Head
Bastion Point
Gabo Island
Cape Howe
Black Head
Green Cape
DISASTER BAY
Mowarry Point
Lennards Island
TWOFOLD BAY
Haycock Point
Merimbula Point
Tura Head
Bournda Island
Mt Delegate
Mt Tennyson
Mt Canterbury
Brown Mountain
Mt Jersey
Mt Ellery
Mt Jack
Mt Puggaree
Jungle Hill
Mt Cann
Mt Raymond
Mt Everard
Lomond Hill
Mt Drummer
Genoa Peak
Mt Kaye
Mt Buckle
Mt Imlay
Mt Poole
Nungatta Mtn
Yambulla Peak
Woolimma Mtn
Mt Carlyle
Mt Nadgee
Mt Buck
Mt Pinnak
Mt Deddick
Mt Bowen
Mt Stradbroke
Mt Tingaringy
Mt Buckland
Bens Peak
Black Mountain
Mt Nunniong
Mt Statham
Mt Dawson
Stringer Knob
Mt Tara
Mt Nowa Nowa
Cape Conran
Sydenham Inlet
Tamboon Inlet
Lake Curlip
Lake Corringle
Mallacoota Inlet
Lake Barracoota
Howe H
Haycock Hill
BONANG HIGHWAY
PRINCES HIGHWAY
MONARO HWY



Map 1a - Ecological Vegetation Classes as mapped in the study region

4.2.1 Ecological vegetation classes

The EVC100 dataset maps vegetation classes defined for the region, at a scale of 1:100 000. Forty one ecological vegetation classes have been mapped for the study region, as shown in Map 1a, and listed in Table 2 on page 48.

As described in chapter 2, ecological vegetation classes (EVCs) are the product of a vegetation typology developed for Victoria by DNRE for use in conservation planning at a regional scale. It was developed in the early 1990s to assess representation at a landscape scale for the old-growth study (Woodgate et al., 1994) and was subsequently used in the East Gippsland Regional Forest Agreement (RFA). EVCs are described as the highest (most general) level in the hierarchy of vegetation classifications used by DNRE (Woodgate et al., 1994).

Accuracy

The metadata supplied with EVC100 gives a positional accuracy of EVC boundaries of 100m to 1km, and attribute accuracy based on attribute checking procedures, as follows:

- polygons > 4ha: an error of 1 in 50 (approximately 98% accuracy)
- polygons < 4ha: an error of 1 in 20 (approximately 95% accuracy)

Methodology

Each EVC consists of one or more floristic communities, which were defined directly from an analysis of the similarity of the species composition at flora survey quadrats. PATN (Belbin, 1995) was used to conduct a non-hierarchical clustering procedure to define groups of sites with similar species composition, which were aggregated to define floristic communities which share a common core of species.

The EVC method in Victoria aggregates floristic communities to define a still more general class. The aim was to go beyond the level of floristic groupings, to create a functional classification. EVCs are based on common characteristics, including floristics, but also life-form, reproductive strategy and physical environment. In this way, for example, cool temperate rainforest in the Otways could be grouped with cool temperate rainforest in East Gippsland, even though they may each share more species in common with the wet eucalypt forest which they adjoin than with each other (RFA Steering Committee, 1996b).

In addition to the site-based floristic data, the sources used to derive EVCs and map their extent were forest type maps and physical attributes including aspect, elevation, gradient, geology, soils, rainfall and salinity (RFA Steering Committee, 1996b).

Questions about Ecological Vegetation Classes as used in Victoria

A number of subjective, expert decisions need to be made to set the parameters which define a classification. In the case of Ecological Vegetation Classes (EVCs) used in Victoria, however, the level of documentation of these decisions limits a full understanding of the basis for classification (Burgman et al., 1996).

This is because although the floristic classification described above is now a relatively standard procedure, the way in which the floristic communities were combined with other data sources to create mapped classes is not made explicit in the published methodologies.

EVC boundaries were defined using API, field checking (generally from roads), and 'reference to mapping of related attributes [such as] geology, landform, topography, climate and forest types' (RFA Steering Committee, 1996, p. g13). The relative importance of these sources of information is hard to determine from the two published methodologies (Woodgate et al., 1994; RFA Steering Committee, 1996b). One source suggests that the polygons are derived largely from an earlier forest type map (DNRE, 1996a). To the extent that the latter is the case, the classification may be primarily representative of variation in structure and canopy tree species. The differences between using floristic and structural criteria to define vegetation classes have previously been an issue of contention in Victoria, over definitions of rainforest (Gell and Mercer, 1992; Rosauer, 1993).

The flexibility in interpolation from site-based floristic communities to vegetation class polygons is expressed in the methodology used for East Gippsland as follows:

> "The delineation and characterisation of Ecological Vegetation Classes is clearly a pragmatic process, making use of whatever of the datasets described above are available and are considered a priori to be likely to reflect the ecological responses of the vegetation to the environmental attributes and the usual disturbance regime of its habitat. The attributes used for characterisation, and the relative influence they have on subjective judgements vary from one EVC to another. ... Mapping, whether it be of vegetation, forest types, land systems or soils, is essentially a craft which requires the skilful and subjective blending of many inputs in an effort to interpolate the distribution of defined entities."(RFA Steering Committee, 1996b), p. g11

The published methodology for the NSW forest ecosystem classification (Keith and Bedward, 1998) also contains undocumented expert decisions, but gives a clear description of the quantitative decision tree structure in which they were embedded, making the process by which the classes were derived much easier to comprehend.

As part of the East Gippsland RFA process, an expert panel reviewed the EVC methodology (Burgman et al., 1996). They were supportive of the general approach, and the use of EVCs for conservation planning at a regional scale, but were concerned at the high level of heterogeneity within four of the classes. Two examples given are the inclusion of dominants with different fire responses within the same class (Burgman et al., 1996), and placing of areas dominated by both *Eucalyptus nitens* and *E. regnans* in the same class, wet forest (Kirkpatrick, 1998).

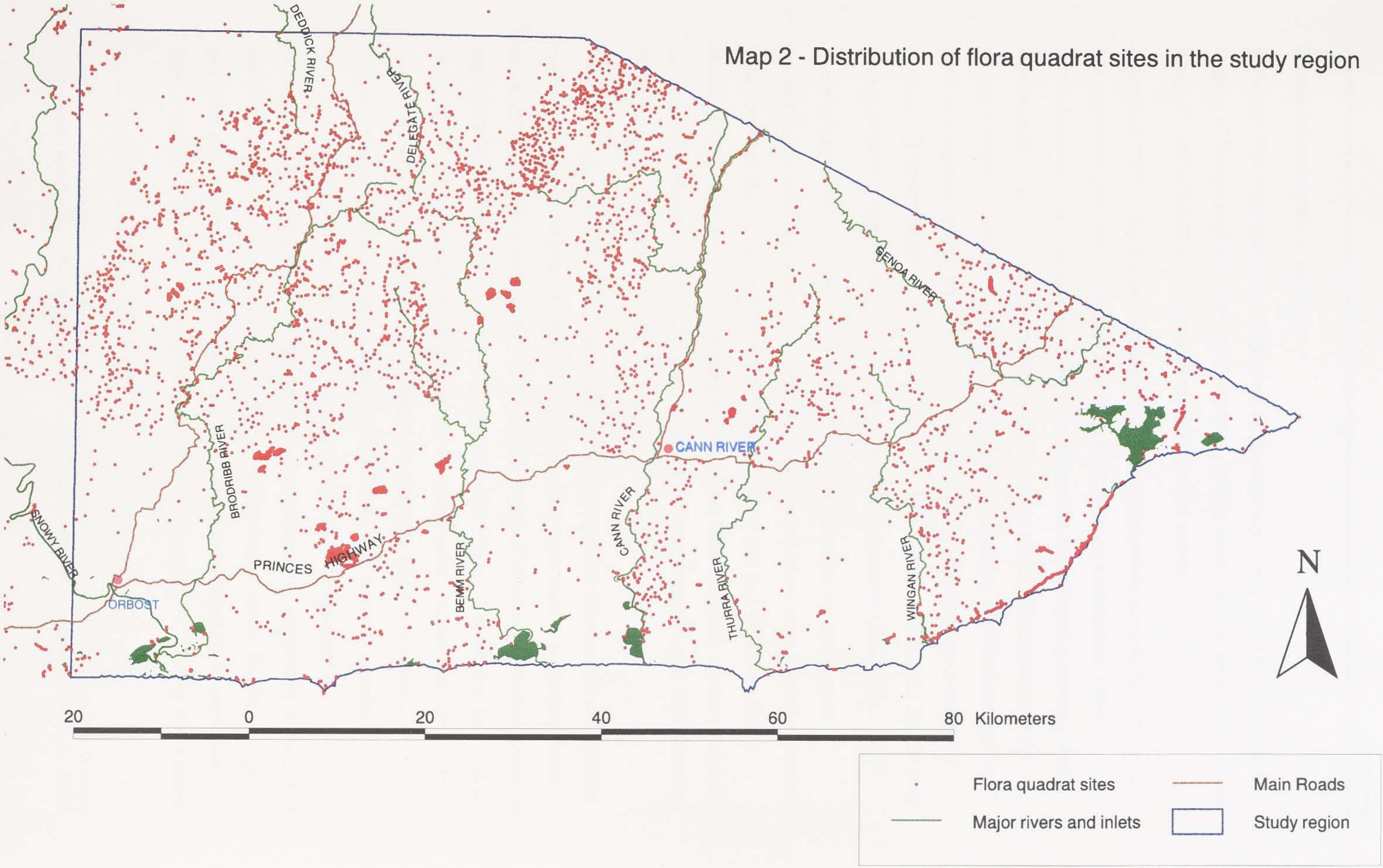
To provide some context, the range of coastal wetlands, heaths and woodlands, for example, together cover less than 2% of the region but are divided into 13 classes. Most EVCs include only one floristic community within the region, although they may encompass others across Victoria (RFA Steering Committee, 1996b). The four broad classes which concerned the expert panel (damp forest, wet forest, lowland forest and shrubby dry forest), however, each include multiple floristic communities, and together occupy 65% of East Gippsland. It is likely that they include well over 65% of the region's timber production. Although the methodology does not provide a reason for their breadth, one explanation is that they were defined broadly to allow maximum flexibility when locating reserves within areas of importance for timber production.

The expert panel recommended that these classes could be more narrowly defined based on their component floristic communities. These classes are well sampled by flora quadrats, so a finer division on the basis of floristic composition should be possible.

4.2.2 Flora survey

The flora data used in this project is a subset of the flora occurrence database maintained by DNRE's Flora Branch. The database includes records from a number of systematic quadrat abundance surveys conducted by Flora Branch and private contractors, as well as herbarium records and species lists for defined regions. For this project only the quadrat data were used. The distribution of quadrat sites is shown on Map 2.

Map 2 - Distribution of flora quadrat sites in the study region



Distribution and density of sampling

Although East Gippsland has a high level of flora survey compared to other forested regions of Australia (RFA Steering Committee, 1996a; Ferrier and Watson, 1997) there is significant variation in sampling intensity across the region.

Individual studies stratified sites within their area to adequately sample environmental and vegetation gradients (Lobert et al., 1991; RFA Steering Committee, 1996b). Compilation of data from many studies means however, that sampling intensity varies significantly across both environmental and geographic space. Descriptions of the survey methods (Lobert et al., 1991; RFA Steering Committee, 1996b) state that core areas of a vegetation type were selected for quadrat sites, in preference to sampling ecotones. This approach would facilitate floristic classification (section 2.1.2), but may miss some rare species which occupy habitat specific to ecotones (Elith et al., 1998).

The most intensively surveyed areas are in high rainfall mountain and plateau environments, while the lowest site density is in the rainshadow areas toward the north-west corner of the region, including the Deddick Valley and part of Snowy River National Park. The distribution of sample sites is also biased towards timber production areas, because much of the sampling was conducted as pre-logging forest block surveys (RFA Steering Committee, 1996a).[2]

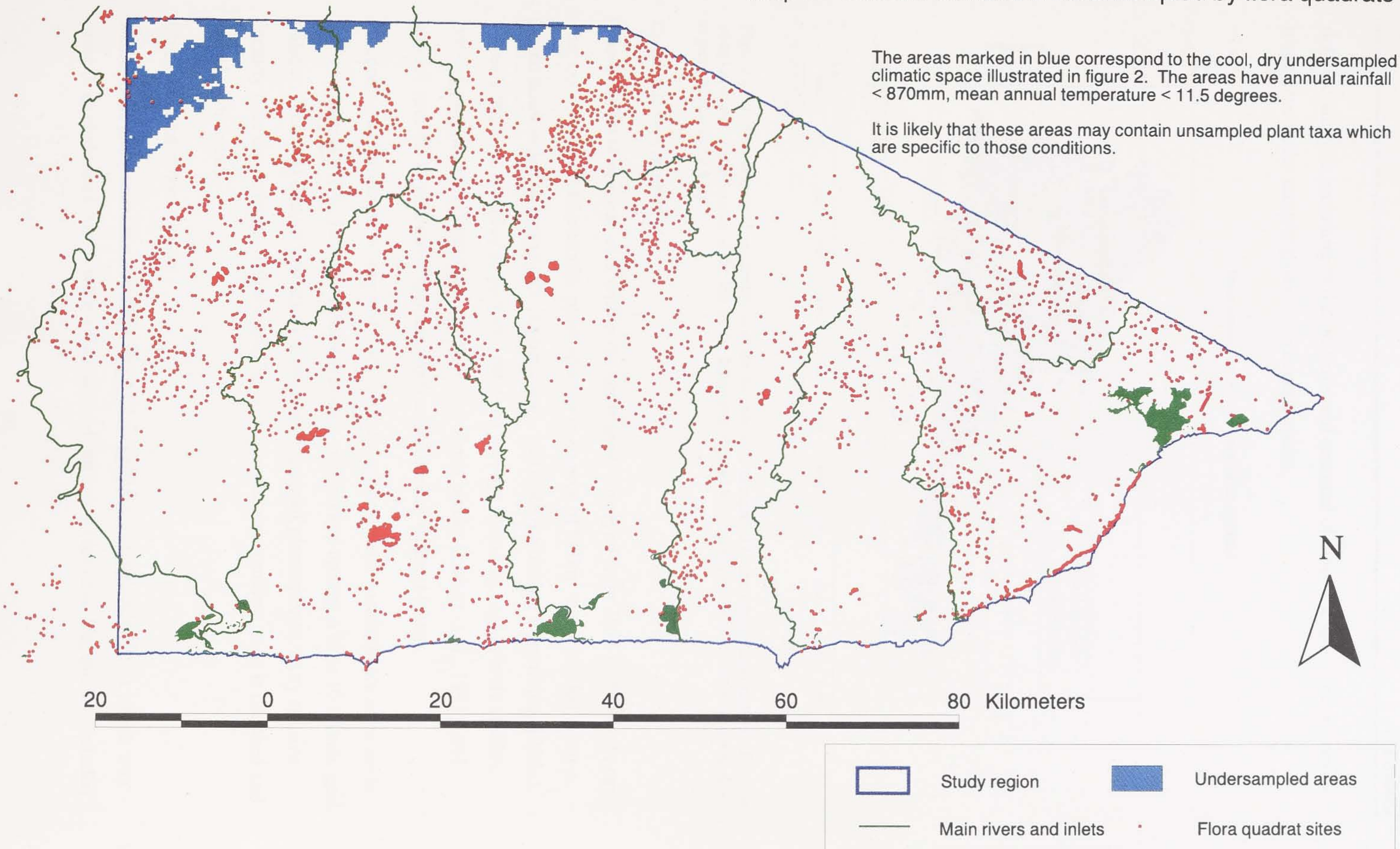
An analysis was performed to determine how well the quadrat sites sample the climatic variation within the region. Using climate surfaces for mean annual temperature and mean annual rainfall generated by BIOCLIM, the range of climatic conditions across the region was compared to the conditions at quadrat sites. Figure 2 illustrates the climatic range of the region, as well as the distribution of sample points within that range. It shows that while most of the range of annual precipitation and temperature are well sampled, the cooler drier areas (<870mm, <11.5°C) are only sparsely sampled. The areas affected are illustrated in Map 3. The significance of this analysis is that it distinguishes sampling gaps in geographic space (which may be covered by another area with similar conditions) from sampling gaps in environmental space. Further sampling to fill the gaps in environmental space may record species or associations particular to those conditions, and thus not adequately sampled in the region. It is likely that a more

[2] The block surveys focussed on logging areas, but extend beyond forest types used for logging. Within a block (approximately 10,000ha) the survey aims to cover the full range of vegetation (Lobert et al., 1991).

Map 3 - Climatic conditions undersampled by flora quadrats

The areas marked in blue correspond to the cool, dry undersampled climatic space illustrated in figure 2. The areas have annual rainfall < 870mm, mean annual temperature < 11.5 degrees.

It is likely that these areas may contain unsampled plant taxa which are specific to those conditions.



detailed analysis including radiation, soils and seasonal climatic variation, would reveal more gaps in the sampling of environmental space.

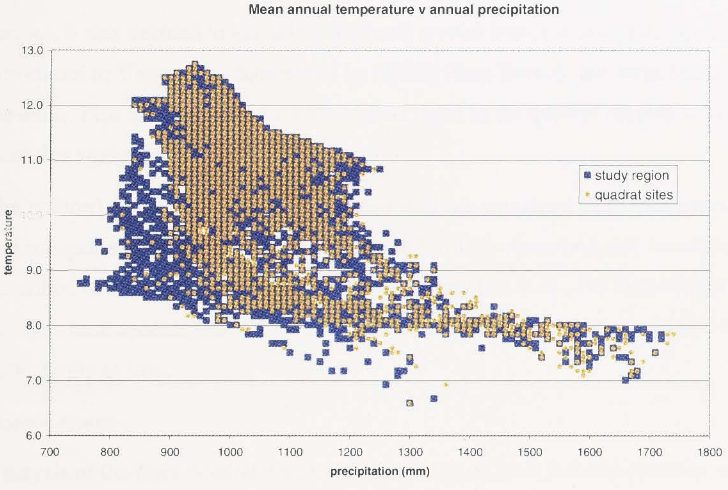


Figure 2 - Temperature and precipitation at the sample sites compared to the whole study region. The less sampled lower rainfall areas at the left of the graph correspond to the areas at the north-west of the study region, as shown in Map 3.

Quadrat survey method

There are 6030 quadrat sites within the study area, surveyed from 1975 to 1998. At each site, occurrence of all vascular plant species in an area of $900m^2$ was recorded using a seven point index of abundance (Lobert et al., 1991; RFA Steering Committee, 1996a). These sites were sampled as part of a range of surveys including region-wide studies, pre-logging flora and fauna surveys based on forest blocks (Lobert et al., 1991), and studies specific to particular habitats, such as rainforests or heathlands.

According to information supplied with the data, quadrat point locations are accurate to within 100m (DNRE, 1996b). Species records are checked against a base 10 minute grid dataset. The grid dataset contains known occurrences of plant species. Any species locality data inconsistent with the grid information and surrounding grids is checked and may be corrected.

Characteristics of flora dataset

From these sites, there are a total of 283 548 records for 2310 plant taxa. A check was done to exclude taxa which were either unclear, mistakes (for example, the list included several records for horse manure and rock), or taxa which were not sufficiently described. A significant number of records were described only to the genus level. Because the site data were used to identify locations for in-situ conservation of plant species, it was decided to exclude introduced species from the analysis. Species introduced to Victoria, as determined by DNRE Flora Branch, are identified as such in the data. Two hundred and one such species listed in the quadrat records were excluded from this study.

The reduced dataset used in the subsequent analysis contained 1226 indigenous species and subspecies The sub-species included are formally described, and accord to the taxa described in the volumes of *Flora of Victoria* (Entwisle et al., 1993) published to date. Because both species and subspecies are included, they are referred to in this paper collectively as taxa.

Floristic diversity

Analysis of the flora quadrat data reveals a picture of great floristic diversity within the study region. This highlights the importance of applying appropriate conservation measures to ensure this diversity is maintained, but also poses a great challenge for sampling the distribution of the many plant taxa, and including them in reserves where required.

As one might expect given previous studies of abundance and rarity (Preston, 1962; Rey Benayas et al., 1999) most of the 1226 indigenous plant taxa recorded at quadrats are uncommon in the sample. This is shown in Figures 3a and 3b. Eleven percent of taxa occur only once, 40% occurred less than ten times, and a mere 9% were recorded at more than one in ten quadrats. One could thus say that rarity is the norm, rather than the exception.

Species accumulation curves (Colwell and Coddington, 1994; Ferrier and Watson, 1997) were used to assess the adequacy of the sample in capturing the full complement of plant taxa in the region. A species accumulation curve plots the cumulative number of different taxa found as a function of sampling effort. Commonly, the majority of species are found at the first few sites, with subsequent sites adding progressively fewer new species. When the curve approaches an asymptote and subsequent sites only give new records of the same species, this suggests that all species have been found. A new site

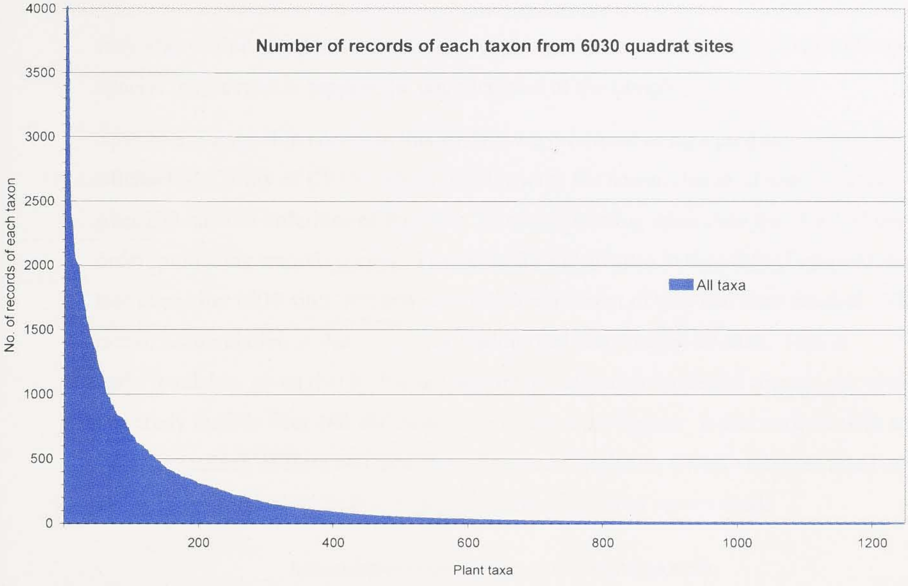


Figure 3a - Number of records of each taxon from the the 6030 quadrat sites. This figure highlights the low abundance of most taxa.

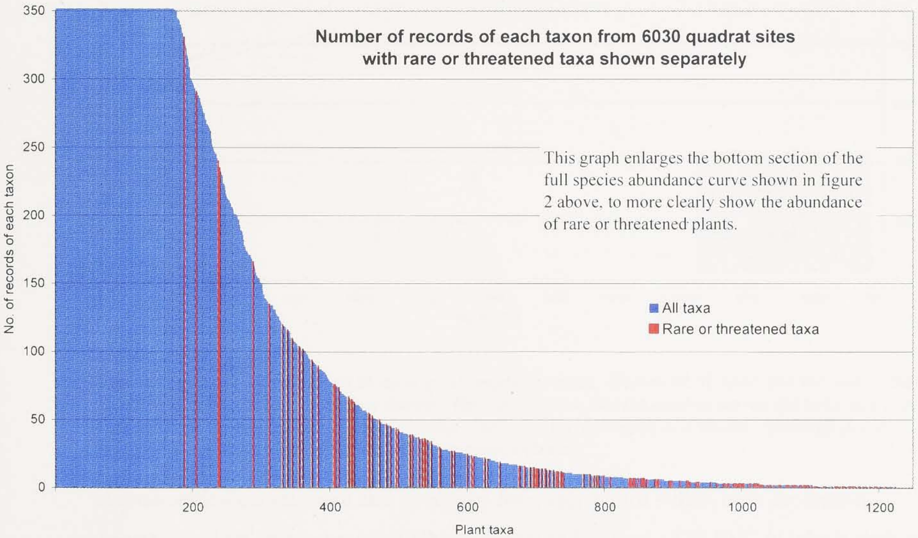


Figure 3b - Number of records of each taxon from 6030 quadrat sites with rare or threatened taxa shown separately. This figure shows that many taxa are rare in the survey data, but are not classified as rare or threatened. For some taxa this would be due to their abundance outside the study area, or records not included in this study.

may always find an additional species, but the existence of many sites which add no new species is evidence to support the completeness of the sample.

Species accumulation curves in this study were produced using a program written by Michael McCarthy of CRES, ANU, which counts the accumulation of species, averaged over 100 random orderings of the sites. Random ordering minimises the effect of site order, producing smooth curves. The curve for the all sites in the study (Figure 4) shows that even after 6030 sites, not quite the full complement of taxa had been reached. The rate of accumulation at that point was 1 additional taxon every 40 sites. This is understandable, given that herbarium specimens and other incidental records not used for this study include over 100 additional taxa in the study region. It also accords with an extensive survey of flora on Cape York Peninsula (Neldner, 1996) which recorded only 65% of the flora known in that region, despite six years of survey work.

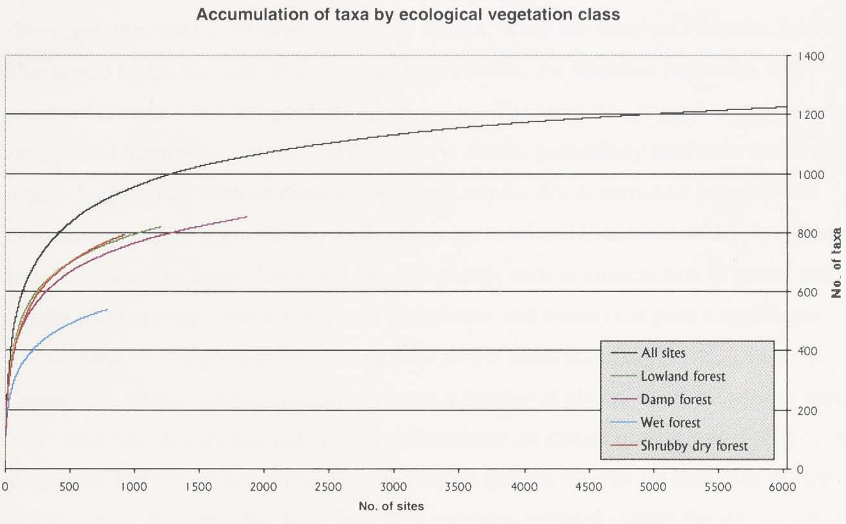


Figure 4 - Accumulation of taxa by ecological vegetation class. Curves for all sites, and for each of the four broadest vegetation classes are shown. The curves show that the quadrat survey did not come close to finding all of the taxa in any of the vegetation classes, or for the region as a whole. Although not plotted here, the same was true for all of the other vegetation classes.

Accumulation curves were also produced for quadrat sites within each ecological vegetation class, examples of which are shown in Figure 4. The EVC in which each site occurred was identified using a simple ArcView intersection of the sites and EVC layers. The results obtained by this method should be treated with some caution, because the range of positional error in both quadrat locations and EVC boundaries means that sites occurring close to EVC boundaries may be allocated to the wrong class.

This analysis is worth reporting despite its associated uncertainty, because it showed that none of the curves for sites within an EVC reached an asymptote, even after as many as 1869 sites (damp forest). This accords with Neldner's (1996) findings, but contrasts with the work of Richards et al. (1990) in the south-east forests of NSW and East Gippsland (including much of the area of the current study), which found curves within an environmental domain quickly reaching an asymptote after 10 to 20 sites. This contrast is surprising. It could result from an element of error in allocating sites to EVCs as described above, or more significantly, from the difference in effectiveness of the classifications used. Richards et al. (1990) used abiotic environmental domains.

Conservation status

Of the 1226 taxa recorded at quadrats in the study region, 187 were listed by DNRE as either rare, threatened, vulnerable or poorly known. They are listed on Victorian Rare or Threatened Plants list, and often referred to as VROTs. As shown in Figure 3b, they occur at between 1 and 331 quadrats in the region. The processes or sensitivities, which have placed these plants on the VROT list vary. Some, particularly rainforest dwellers, require long periods without disturbance, others require fire at particular intervals and intensities to trigger regeneration, while others are restricted to habitats which have been greatly reduced in extent. Localised site disturbance such as construction of gravel pits, or new logging roads (which bring both disturbance and weeds) can pose a significant risk to many taxa, especially where they exist in few small populations (RFA Steering Committee, 1996). Understandably, there are a number of plants, which because of their rarity have not been studied sufficiently to determine the causes of their rarity. Figure 3b also shows that not all taxa which were rare in the quadrat records are classified as rare or threatened. For some this is because they are more common outside the region, but it would be worth investigating whether some ought to be on the VROT list, but have been overlooked.

While recognising that the majority of these taxa will benefit from being in areas managed primarily for conservation, this broad scale study does not consider their individual needs. Rare or threatened taxa are treated as a group, whose rarity, vulnerability and scattered distributions pose a challenge for reserve design, and thus require particular attention.

4.3 The reserve design tool Spexan and its application in this study

This project uses a new reserve selection algorithm called Spexan, designed by Ian Ball of Adelaide University, in collaboration with Environment Australia's Forest Taskforce. It has been used to date on a trial basis in development of reserve options for Regional Forest Agreements. Spexan includes features which are particularly useful for reserve design, and led to its use for this project, in particular its ability to handle very complex problems, including consideration of cost and spatial configuration. In this section, the operation of Spexan is described, along with the processes and choices involved in applying it to the study region.

4.3.1 Overview of Spexan

Spexan selects reserve areas in order to simultaneously optimise for three factors:

- representation of conservation values within reserved areas;
- cost of the reserve system
- spatial configuration of the reserve system

The selection process aims to meet conservation goals regarding the minimum amount of particular mapped attributes to include within reserves, while maximising the degree of adjacency and compactness of reserve areas, and minimising the cost of the reserve system. These three objectives are partially incompatible, so each objective limits achievement of the other two. The extent to which each of these goals determines the areas selected, is dependent on weightings placed on them.

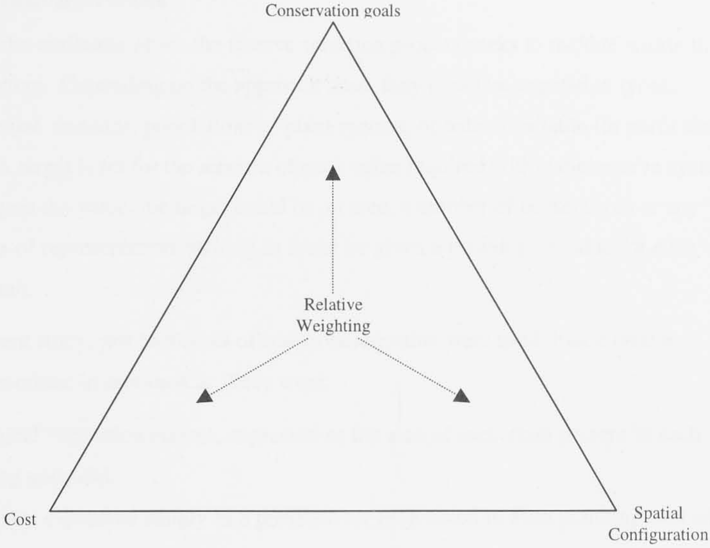


Figure - A potential reserve system generated by Spexan involves a trade-off between a) meeting conservation goals by inclusion of specified attributes within reserves; b) spatially configuring reserve areas to assist ecological and population viability; and c) minimising the cost of the reserve system. The weighting of these factors alters their relative influence on the choice of reserve areas.

The unit of selection which the selection algorithm chooses is called a planning unit (PU). These are defined areas of land within the study region, which can each be either part or not part of a potential reserve system. A database stores information about the attributes of each planning unit.

The reserve selection algorithm has two components:

- An objective function which measures how well the reserve system meets its objectives. The lower the value of the objective function, the closer the reserve system is to meeting its objectives;
- A search routine which uses simulated annealing to seek the set of reserved planning units which gives the lowest value of the objective function.

The following sections describe each of these elements in more detail, and then explain how it was used in the East Gippsland case study.

4.3.2 Conservation values

These are the attributes which the reserve selection process seeks to include within the reserve system. Depending on the approach used, they could be vegetation types, environmental domains, populations of plant species, or habitat suitable for particular animals. A target is set for the amount of each value required within the reserve system. Depending on the value, the target could be an area, a number of occurrences or any other index of representation, so long as it can be given a meaningful value for each planning unit.

In the current study, just two types of conservation value were used, based on the datasets described in section 4.2. They were:

- Ecological vegetation classes, expressed as the area of each class present in each planning unit; and
- Plant taxa, expressed simply as a presence for each taxon in each planning unit where it was recorded at a quadrat site.

Targets were set for the representation of each vegetation class and plant taxon.

Targets for representation of Ecological Vegetation Classes

Targets for the inclusion of each EVC within the reserve system were based on the estimated pre-European extent of that class. This approach has been used in the RFA process, and the rationale in this project was the same. That is, that where the extent of vegetation class has been changed since European settlement by clearing, logging, changed fire regimes or some other effect, this should not result in less of what remains being protected. The pre-European areas used in this study were based on the estimates published in the East Gippsland CRA Environment and Heritage Report (RFA Steering Committee, 1996a) which also briefly describes methods and estimates of reliability for these areas.

Various reserve representation targets could be justified as a necessary compromise between conservation and resource use. Ideally the implications of a range of target levels would be analysed and compared. For this project it was only possible to consider one target level. A target of 10% representation of each EVC was used, which accords with the recommendation of the IUCN (IUCN, 1993) and to a similar study undertaken in South Africa (Lombard et al., 1997). Table 2 below shows the pre-European and 1996 areas of each EVC within the study area, and the resulting reservation targets.

Table 2 - Current and estimated pre-European areas of Ecological Vegetation Classes in the study region, and the resulting reserve representation targets in hectares. The resulting target for all classes is 9.9% of the region, but 10.7% of the public land.

Ecological Vegetation Class	Pre-European area (ha)	Current area (ha)	Target used (ha)	% of current area required
Coastal Dune Scrub Complex	2770	2659	277	10%
Coastal Banksia Woodland	3060	3099	306	10%
Coastal Grassy Forest	680	6	6	100%
Coastal Vine-rich forest	90	137	9	7%
Coastal Sand Heathland	660	663	66	10%
Clay Heathland	1030	649	103	16%
Wet Heathland	10000	9602	1003	10%
Coast Saltmarsh	590	1343	59	4%
Estuarine Wetland	1090	504	109	22%
Coastal Lagoon Wetland	820	853	82	10%
Wet Swale Herbland	4	4	1	25%
Brackish Sedgeland	190	195	19	10%
Banksia Woodland	39340	36712	3934	11%
Limestone Box Forest	470	46	46	100%
Lowland Forest	216940	200649	21694	11%
Riparian Scrub Complex	19580	17305	1958	11%
Riparian Forest	21080	12337	2108	17%
Heathy Dry Forest	430	2160	43	2%
Shrubby Dry Forest	101560	98750	10156	10%
Grassy Dry Forest	7380	3330	738	22%
Box Ironbark Forest	600	596	60	10%
Rain Shadow Woodland	15	15	2	10%
Rocky Outcrop Scrub	720	676	72	11%
Rocky Outcrop Shrubland	740	744	74	10%
Damp Forest	185570	181011	18567	10%
Wet Forest	81310	80617	8131	10%
Cool Temperate Rainforest	2350	2350	235	10%
Warm Temperate Rainforest	10270	6254	1027	16%
Cool/Warm Temperate Rainforest Overlap	220	222	23	10%
Tableland Damp Forest	4880	6680	488	7%
Montane Dry Woodland	11710	7056	1171	17%
Montane Grassy Woodland	3990	54	54	100%
Montane Damp Forest	420	423	42	10%
Montane Wet Forest	5620	5624	562	10%
Montane Riparian Woodland	970	175	119	68%
Montane Riparian Thicket	40	37	4	10%
Sub-alpine Woodland	910	677	90	13%
Treeless Sub-alpine Complex	240	234	24	10%
Herb-rich Forest	8510	5867	851	15%
Dunes	1820	1805	182	10%
Lakes and Inlets	3300	3300	-	
Cleared Private Land	-	58722	-	
Total		754142	74495	9.9%

As Table 2 shows, the 10% target was applied consistently, except where 10% was greater than the total remaining area of that class, in which case the target was reduced to the total current area.

As discussed, the use of an across the board 10% target is an experimental procedure to examine the relationship between EVCs and plant distributions. A real application of this approach would require, at minimum, some basic changes. Firstly, a threshold could be added, so that classes with restricted areas (in the region or more broadly) would not be further reduced. The target could be, for example the full area up to 1000 hectares or a given percentage, whichever is greater. Secondly, some vegetation types, such as montane riparian thicket, coastal lagoon wetland and all rainforest classes, are classified as rare, vulnerable or endangered in Victoria, and already enjoy protection which would need to be recognised in reserve planning.

Targets for representation of individual taxa

The target used for individual taxa, was simply that the reserve system must include at least one planning unit where each taxon had been recorded. This is a simple, but minimal requirement for conservation, which ensures that the reserve system will represent floristic diversity, but does not address issues of population viability.[3] Other studies which have taken a similar approach to species representation include Lombard et al. (1997) and Margules et al. (1988). Use of the same target for all taxa creates a strong bias towards rarely recorded species, which end up greatly influencing the shape of the reserve system. This is because common species meet their target so easily that they put little constraint on the reserve design. In a more comprehensive study individual targets could be set for the areas of habitat required for each taxon, based on factors such as life history, current abundance, endemism, estimates of minimum viable population, and the likelihood of persistence in areas not managed for conservation. One method for setting such targets for a suite of threatened species where information is limited, is described in Burgman et al. (in press).

4.3.3 Value weightings

The value weighting factor modifies how important it is to meet the representation target for each conservation value, that is, for each vegetation class and plant taxon targeted for protection. The aim in this study was to meet the targets for all values, so the value weighting was adjusted upwards until the model consistently met all targets. It would be possible to set the weighting individually for each vegetation class or plant taxon based

[3] An alternative approach would be to use a habitat model to map potential habitat for each taxon, and then set a target for the area of habitat to be reserved. This approach is discussed further in section 4.3.6.

on its degree of importance or vulnerability. In this project however, weightings were only set to differentiate three groups of values: vegetation classes, rare or threatened taxa, and all other taxa, to produce the four strategies described below in section 4.4.

4.3.4 Spatial configuration

Spexan uses the perimeter, or boundary length of the entire reserve system as a simple index of adjacency. By seeking to minimise boundary length, the system has a preference for fewer, large reserves over many small ones, and for reserves shaped to have low perimeter area ratios. A boundary length multiplier can be set to adjust the weighting, or strength of this preference relative to other factors.

4.3.5 Planning units

Planning units are discrete, mapped areas which are the units of selection for the reserve system. They may be defined in a number of ways, for example to represent areas derived from:

- current land management, such as coupes, compartments, blocks or land parcels, or even road boundaries;
- natural features, such as catchments;
- environmental classifications, such as land systems or forest types, or
- a regular grid.

For the current study, in the absence of access to the actual planning boundaries used in the region, it was decided to create a grid of regular planning units to cover the region. The two major considerations were the shape and size of the units.

Shape of planning units

A grid of square planning units is the most obvious and easily implemented in a GIS system. Square selection units were used in similar analyses in South Africa (Lombard et al., 1997) and Australia (RFA process, unpublished) and in the trial run for this project.

A problem with using Spexan with square PUs, is that because the algorithm assesses adjacency in terms of shared boundaries, areas which meet only at a corner share no boundary, and are thus treated the same as widely separated areas (see Figure 6). Lombard et al. (1997) used a different system to consider adjacency, and incorporated separate rules to deal with squares touching along sides and corners. The solution developed in this project was to use hexagons. Unlike other regular polygons, hexagons avoid the problem of units touching at a corner, because they form a grid in which any two hexagons that touch do so along a shared face.

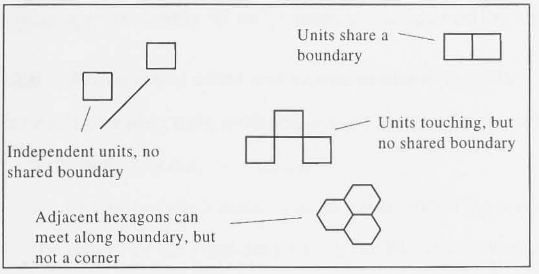


Figure 6 – Adjacency in square and hexagonal grids. Spexan uses shared boundary as a measure of adjacency. This creates a problem for the use of square planning units, which are adjacent, but share no boundary. Hexagons avoid this problem because any hexagons which touch have a shared boundary.

Size of planning units

For results to be meaningful, the size of planning units needs to be appropriate to the spatial scale of the environmental data used in the project. There is no point in using units which divide the landscape more finely than was done in collection or classification of the input data. Larger PUs make allowance for positional error (eg for EVC boundaries) and help to smooth differences in point sampling intensity (Kirkpatrick and Brown, 1994).

The smaller the PUs used, the finer the scale at which areas required within the reserve system can be selected from those which are not. So long as this is justified by the scale of the input data, the resulting reserves more closely follow the ecological boundaries which guide their selection, and include less of unwanted elements. Reserves designed with smaller PUs thus tend to be more efficient (Pressey and Logan, 1998). Another consideration, is that the more planning units used, the larger the computing power, data storage and processing time required.

The size chosen as a compromise between these factors resulted in a planning unit area of approximately 92ha.

Generation of planning units

The coordinates for a field of hexagons were generated using a program written for the purpose by Julie Clutterbuck of the Maths Department, ANU. These coordinates were fed into ArcInfo to generate a polygon coverage, and clipped to follow the borders of the region. PUs at the edges of the region are thus not hexagonal, but instead follow the coastline or region boundary.

The result of this process is a field of 8385 planning units covering the region. Each PU covers approximately 92 ha[4], except at the edge of the region where size is variable.

4.3.6 Allocation of costs and values to planning units

For each planning unit, a database used by Spexan records the following attributes:

- a measure of cost;
- the amount of each conservation value occurring within the PU;
- the length of the boundary which the PU shares with each adjoining PU; and
- availability for reservation (available, not available, pre-determined as reserved).

For this project, planning unit area was used as a simple surrogate for the cost of each PU, which was equivalent to its area in hectares. This means that although the model sought the smallest area which met the conservation targets, taking into account variation in the size of planning units, it did not place a higher cost on areas with potential value for timber, minerals or other uses. In this regard, the approach used was similar to other reserve selection techniques based on minimum area. The area of each planning unit was extracted from ArcView, and used to create a cost database.

The two types of conservation values (section 4.3.2) were allocated to each planning unit. Firstly, the area of each EVC occurring within each PU was determined using a simple intersection in ArcView.

Secondly, a similar procedure was used to determine which flora quadrat sites occurred in each PU. This information was loaded into a database containing the records of the plant taxa identified at each quadrat site. The database was queried to produce a list of plants recorded in each PU.

To illustrate this process, Map 3a shows a section of the study region with EVCs, quadrat locations and planning units.

The chance of a plant taxon being recorded in a PU depends, in part, on the number of quadrats included in the area. A significant number of PUs contained no quadrat sites,

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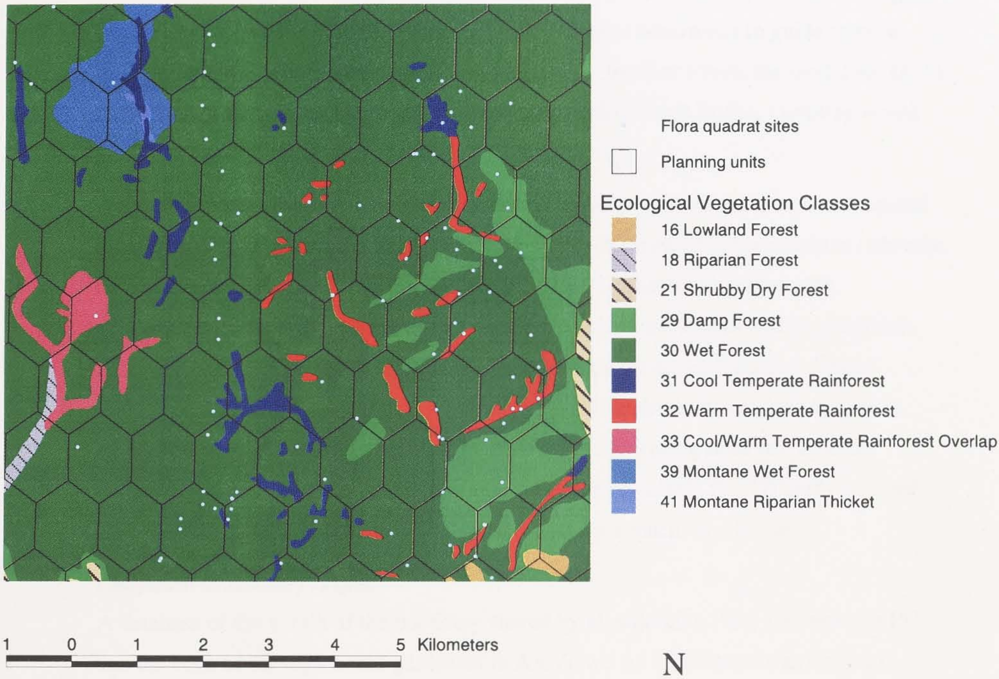
[4] The planning units were generated in a geographic map projection, as hexagons with a side length of 0.006 degrees. Conversion to a transverse Mercator projection (UTM) resulted in a variation in planning unit area with latitude. The variation in area (92ha ± 0.7ha) across the region is not significant for the current study.

Map 3a - Allocating attributes to planning units



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This map shows Ecological Vegetation Classes and quadrat sites in part of the upper Goolengook catchment. Planning units (PUs) are included to illustrate how attributes were recorded for each PU.

A database used by Spexan recorded the area of each EVC occuring in each PU, and all the taxa found at flora quadrats within the PU.

and thus would not be reserved for the plant species that occur in them. This means that in those of the models described below that used species occurrence to guide reserve design, there was a bias towards well sampled areas. In other words, the model sought to reserve plant taxa where they have been recorded, even though further sampling would yield more occurrences of many taxa.

An alternative approach to listing the taxa in each planning unit, would be to use spatial habitat models to predict the likely distribution of each taxon from the presence / absence data in the quadrat survey (Ferrier and Watson, 1997). This would reduce the dependence of the model on the exact location of quadrat sites, both within and across PUs. This approach was not taken in the current study for several reasons. These included the time and data storage required to model hundreds of taxa, and the errors which would be introduced to models by deriving models using flora survey data covering only part of the range of most taxa. The data available for this study covered East Gippsland only, while most taxa in the area have a much broader range.

Calculation of boundary lengths

A database of the length of the boundary shared by all adjoining PUs, and between PUs and the edge of the region was generated in ArcView. An adjustment was made to reduce the length recorded for boundaries along the coast to reflect the fact that a coastline is a natural ecological boundary, not a cause of new fragmentation. There is no reason to minimise the length of coastline in reserves, in contrast to minimising boundaries with areas used for cultivation, grazing or intensive logging.

4.3.7 The reserve selection algorithm

As introduced on page 46, the reserve selection algorithm has two components, an objective function and a search routine.

The objective function

The objective function measures how good a given reserve configuration is. Specifically, it evaluates how far a given selection of planning units is from meeting the objectives defined for the reserve system. The lower the objective function, the better the reserve system meets its objectives.

These objectives are to:

- include the target amount of each conservation value in reserved areas

- minimise the total length of the boundary between reserved and non-reserved areas
- minimise the aggregate cost of the areas reserved

The objective function is calculated according to the formula:

$$\sum_{PU} \text{Cost} + \sum_{PU} \text{Boundary} \times \text{BLM} + \sum_{Value} (\text{Penalty} \times \text{ValueWeighting})$$

where:

$\sum_{PU}$ Cost is the total cost all planning units within the reserve system.

$\sum_{PU}$ Boundary is the length of the boundary separating planning units within the reserve system from those which are not.

BLM is a boundary length multiplier to adjust the weighting of boundary length relative to cost and target penalties. It can be adjusted for different units of measurement, or to vary the priority placed on compact reserve configuration.

$\sum_{Value} (\text{Penalty} \times \text{ValueWeighting})$ is the sum of penalties for the amount by which each conservation value falls short of its target. The penalty for each value is multiplied by the weighting for that value. A highly weighted value would thus attract a greater penalty for falling short of its reservation target. As all targets are achieved or exceeded, the last term would reduce to 0.

The reserve design process seeks the set of planning units which results in the lowest value for the objective function. This means, for example that if two configurations both include target levels of all conservation values, but one occupies less costly land, or has a shorter external boundary, it will be preferred.

The next section describes how Spexan seeks to minimise the objective function, and thus find the set of reserve areas which best meet the objectives for representation, cost and spatial configuration.

4.3.8 The simulated annealing search routine

Spexan searches the solution space by repeatedly adding or removing random planning units from a potential reserve for a specified number of iterations. The value of the objective function is recalculated after each change. Changes which lower the objective function and bring the reserve system closer to its goal, are retained. The solution reached after many iterations may not be the best possible solution, but will be a local minimum.

As mentioned in chapter 2, Spexan uses simulated annealing, an optimisation technique drawn from operations research (Kirkpatrick, S. et al. 1983). It was developed to solve a problem inherent in iterative improvement techniques involving many independent variables. The problem is that a solution may often be found which appears to be optimal because it cannot be improved by making any one change, but is in fact only a locally optimal solution within the space of possible solutions. There may be a whole range of better solutions which would be missed, because they require the addition or removal of several areas which each actually make the solution worse, before it begins to improve again.

To deal with this problem, simulated annealing changes the criteria for deciding which changes to accept. Instead of only accepting changes which reduce the objective function, it also allows changes which actually increase the objective function by up to a certain amount, enabling it to escape from local minima as illustrated in Figure 7. This amount is called the temperature (Ball, 1998).

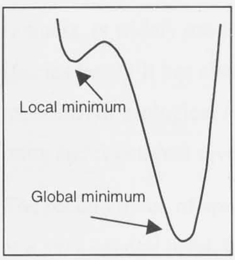


Figure 7 - Global and local minima for a hypothetical function. The temperature in a simulated annealing algorithm helps it to escape from local minima in the multi-dimensional solution space of the problem, represented here in simplified form.

A Spexan run starts with the temperature set high, so most changes, good or bad, are accepted with little restriction, or chance of getting stuck. As the iterations proceed the temperature gradually cools, increasingly restricting changes to those which move the system closer to its objective. When the temperature reaches 0, the algorithm allows only changes which directly reduce the objective function.

The name simulated annealing is used because of the analogy to the gradual cooling of a metal which enables it to settle into a stable crystalline structure, where rapid cooling (reaching a solution too quickly) would cause it to be brittle, having frozen too quickly, before it settles into a stable configuration (low value of the objective function).

Another analogy which may help to understand simulated annealing, is of a ball rolling across a mathematical surface created by solutions to objective function. The ball rolls downhill, towards the lowest point, or minimum value of the function. This would correspond to the selection of reserve areas which best meet the reserve objectives. The ball may however become trapped in a depression which is far from the lowest point. The temperature described above, gives the ball momentum which enables it to roll out of the local minimum, and probably find a better solution.

4.3.9 Measures of reserve performance

Four measures of reserve performance were used to assess and compare the different reserve design strategies.

Spatially, the reserve systems were described by their total area in hectares, and their perimeter - area ratio. The latter measure quantitatively combines the shape of reserve areas and the degree to which reserve areas are small and scattered or few and large.

Two indices of species representativeness of reserved areas were used. The first, species richness, is widely used because it provides a simple indication of biodiversity at the species level. It has also been found to be positively correlated with a range of other measures of biological and ecological diversity, such as diversity of higher taxonomic units and functional diversity (Gaston, 1996a).

The second index of species representativeness was developed for this project. It aims, at a very general level, beyond just counting presence or absence in the reserve system, to indicate how well the taxa are represented. Referred to as the representation index, it is derived by a simple two–step calculation. Firstly, for each taxon in the study, the proportion of planning units with records of the taxon is calculated as follows:

$$RI_i = \frac{\text{no. of planning units where taxon } i \text{ has been recorded within reserves}}{\text{no. of planning units where taxon } i \text{ has been recorded within region}}$$

This number is then averaged across all taxa. The resulting index ranges from 0 to 1. It is consciously biased towards less common taxa, as the following example illustrates:

> Two species were each recorded 5 times within the reserve system. Species A occurs 10 times in the whole study area, while species B occurs 50 times. Species A will have an RI of 0.5 compared to 0.1 for species B.

This index has not yet been carefully evaluated, and subsequent work may well show a better index of representation, however it does provide useful information beyond species richness, on how well a large number of taxa are included in a planned reserve system.

4.4 Analyses and comparisons performed

To address the aims given in chapter 3, modelling was undertaken in two stages. In the first stage, a series of potential reserve systems were created, to achieve a goal of 10% representation of each EVC (strategy A). Over a series of runs the mean area required to achieve this was determined. In the second stage, spatial information for all plant taxa in the study were loaded into the system, and the taxa used as targets to see if better species representation could be achieved within the same total area (strategies B – D).

The four different reserve selection strategies used (A-D), in which the model was run with different conservation goals and data are described below. These four strategies compare the effects of selecting reserve areas based on distributions of either vegetation classes, flora site records, or a combination of both.

- Strategy A required reservation of 10% of the area of each vegetation class. The resulting reserves were assessed for species richness, as described below. The aim was to assess how well reserves designed to be representative of vegetation classes would represent floristic diversity defined by taxa.

  In the subsequent strategies, the area available for reserves was capped at the mean value required in strategy A. The purpose of the subsequent strategies was to compare the result which could be achieved within the same total area, by selecting directly from flora site data, with a specific requirement for inclusion of each species.

- Strategy B required records of rare and threatened plant taxa in the study, as well as 10% of the area of each vegetation class, to be represented within a reserve system of the same total area as in A.

- Strategy C was similar to B, but required records of all plant taxa as well as 10% of the area of each vegetation class within the same total area.

- Strategy D was significantly different, seeking records of all plant taxa in the study, but without any goal for representation of vegetation classes.

All strategies were compared in terms of species representation and spatial configuration.

Because of the stochastic element of the selection algorithm, each outcome produces elements which are unique. The variation between outcomes may be small or very great, depending on how much opportunity there is to meet the same conservation goals by reserving different parts of the region. For this reason, batches of runs were used for each strategy, so that the analysis could consider the range of outcomes generated.

Table 3 – Summary of the four selection strategies used. EVC = Ecological Vegetation Class, VROT = Victorian Rare or Threatened plant taxa.

Strategy	Targets	Area limit	Repeat runs
A	10% of each EVC	None	30
B	10% of each EVC and VROT taxa	Mean area of strategy A	10
C	10% of each EVC and all plant taxa	Mean area of strategy A	10
D	Include all plant taxa	Mean area of strategy A	10

The analyses performed are described in greater through the remainder of this chapter.

4.4.1 Calibrating the model's spatial configuration function

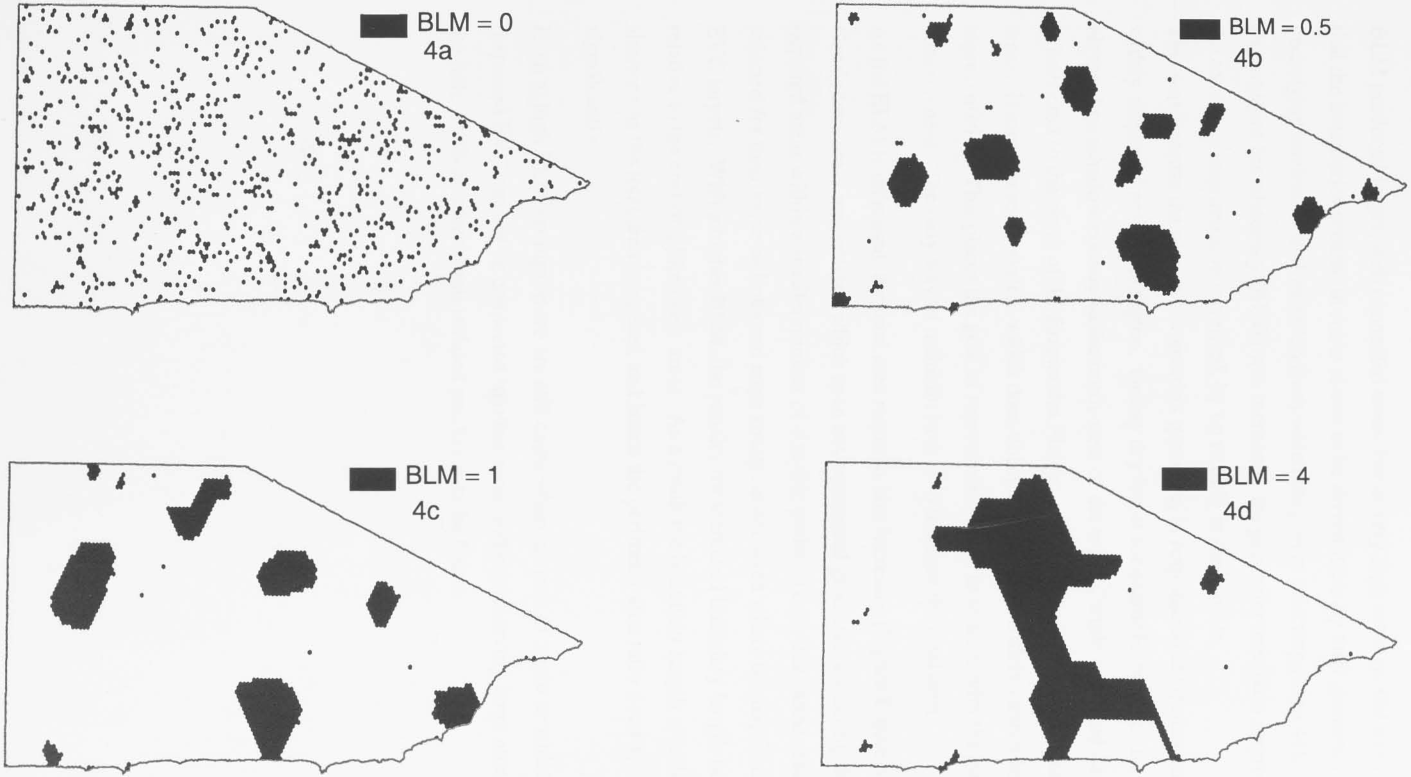
Using the 10% targets for each EVC, the model was first calibrated over a series of test runs.

A series of 5 batches of 10 test runs were performed to calibrate the model's spatial configuration function. Each batch used a different setting of the boundary length multiplier (BLM), to examine its effect on spatial configuration, and determine the best setting to use.

The outcomes illustrated how altering the priority placed on adjacency influences the spatial configuration of reserve areas, their perimeter area ratio, and the total area required to meet reserve targets.

Figures 8 and 9 below, show the averages of ten runs at each BLM value, with reserve cohesiveness increasing on each graph as the BLM increases from left to right. Examples of the reserves generated at each BLM setting are shown in Maps 4a-d. Different as they look, each result meets the same targets for representing the region's vegetation classes. With BLM set to 0, adjacency is not a factor in selection of planning units, so the required areas are dispersed in a seemingly haphazard manner. While this is the most efficient in terms of the total area required, it is likely to be a very poor reserve design. A very low BLM produced many very small, isolated reserve patches. A higher

Maps 4a-d - Calibrating the boundary length multiplier



Maps 4a-d illustrate the different reserve configurations produced with different settings of the boundary length multiplier (BLM).
Each configuration satisfies the same representation target, by including 10% of each vegetation class.

BLM produced larger, well connected areas, but at very high settings, did so to the extent that the boundaries seemed in some places to be driven more by their geometry than the underlying environmental variation from which they were selecting (Map 4d). As the requirement for adjacency (BLM) was increased, the perimeter area ratio improved, but the total area required also increased; by up to 60% in these trials.
This was because the required vegetation types are, in response to environmental drivers, widely dispersed across the region. Grassy dry forest for example, only occurs in elevated rainshadow country in the north-west of the region, while tableland damp forest is restricted to the north of the Errinundra Plateau, and estuarine wetland, to near coastal areas. Hence a reserve system which does not span the region widely cannot be representative. This places the goal of representativeness in tension with the goal of cohesiveness. The only way to maintain both is to increase the total area.

As the BLM is increased, the total area required also increases (Figure Figure 8). Contiguous areas are selected which span environmental gradients, capturing the required areas within a smaller number of sizeable parks. To do this, some areas are selected for their value as links and edge straighteners, even where not required to meet EVC targets. With a higher BLM, the penalty for increased boundary length increases relative to the cost of adding new areas. As a result the boundary length actually shortens as the total area increases, and hence the perimeter area ratio decreases significantly.

Even at high BLM settings, there are still cases where an area of a rare or widely dispersed EVC can not be connected to other areas without reserving large areas just for the link. In these cases small isolated patches may be found.

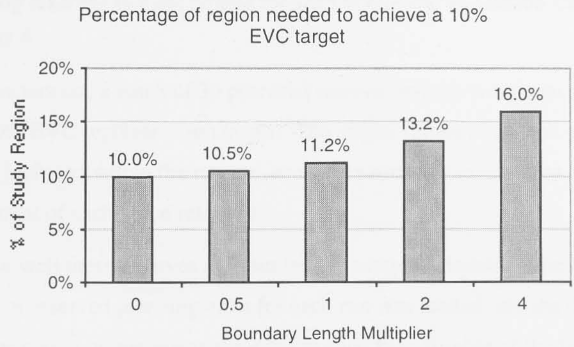


Figure 8 – Percentage of the region needed to achieve a 10% EVC target. This graph gives the average of 10 runs at each of 5 BLM settings. It shows that the greater the priority placed on adjacency and cohesiveness, the larger the total area needed to meet the representation targets. In this case, a scattered reserve system met the same representation targets using 60% less land. If the focus is solely on meeting representation targets, rather than on the adequacy of the reserve system for conservation, such an option could appear attractive.

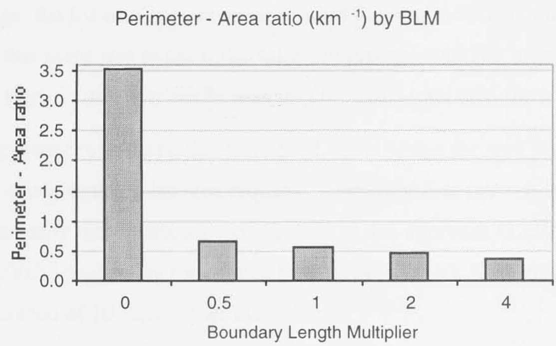


Figure 9 – The effect of varying the priority placed on adjacency – A summary of 10 runs at each of 5 BLM settings. This graph, based on the same reserves as Figure 8 above, shows how increasing the BLM decreases the perimeter-area ratio, resulting in reserves which would be less subject to influences from the management of surrounding, unreserved lands.

From examination of these results, an intermediate BLM value (BLM=1.0) was selected for use in strategy A. Although based on consideration of the results in Figures 8 and 9, and visual inspection of the reserves generated, calibration of the BLM was essentially a subjective judgement.

4.4.2 Selecting reserves to meet targets for each Ecological Vegetation Class - Strategy A

With the parameters set, a batch of 30 potential reserve systems were generated, each meeting the 10% EVC representation target. The output for each run was a list of planning units included within the reserve, as well as statistics on the area, boundary length and amount of each value reserved.

To analyse how well these reserves represented the sampled floristic diversity of the region, the list of reserved planning units for each run was loaded into the database containing information on the plants recorded in each PU. A table of the frequency of occurrence of each of the taxa within the reserve system was produced, and from this, the species richness and representation indices were calculated for all taxa together, and also for just VROT taxa.

4.4.3 Selecting reserves directly from occurrence of plant taxa – Strategies B-D

In the second stage, the list of plants recorded in each PU was included in the model. The challenge at this point was to see if the selection process could be improved to better represent known floristic diversity while selecting the same total area for reserves.

Three different approaches to doing this were tried. In all cases the area available for reservation was limited to the mean area required in strategy A to reserve 10% of each EVC. Because so many more values are considered in this approach (1226 species, compared to 45 EVCs) processing took much longer. As a result, the batch for each set of conditions consisted of 10 runs, rather than 30.

- Strategy B requires 10% of each EVC, as well as requiring all VROT taxa to occur within the reserve system.
- Strategy C is similar to B, requiring 10% of each EVC, but broadens the taxon based target to required all taxa in the study to be recorded within the reserve system. Because the area available in B and C is already required to meet the each of the EVC targets, there is little scope in these strategies to select additional areas of some EVCs to achieve the species targets. This means that strategies B and C effectively asks the system to find the required taxa largely by altering the location but not size of areas chosen within each EVC.

- Strategy D takes a different approach, removing the EVC target altogether. The target in strategy D was simply for all taxa to occur within the reserve. This is a recipe for a reserve system built entirely by selection at the taxon level, with no reference to broader ecological or floristic classifications.

The four strategies can be seen as a continuum, starting at a coarser level of biodiversity, the ecosystem, and graduating to a finer level, the taxon.

5 Results and discussion

This chapter illustrates the reserve options generated by the methods described above. The merits of the different selection strategies are examined, and the strengths and shortcomings of the method as a whole are considered The analyses described above are presented and discussed in three sections. The effectiveness of the four strategies at encompassing the sampled floristic diversity of the region within reserves is considered in Section 5.1. The spatial characteristics of the reserves generated by each of the strategies are shown in Section 5.2, along with some additional analysis of Spexan's spatial configuration function. Although presented separately for clarity, the representativeness and spatial aspects of reserve selection are in no way separate issues. They are drawn together in Section 5.3, which summarises the results.

Shortcomings of the method used are considered both in terms of the limitations they place on interpreting the results of this study, and improvements which could be made for subsequent work.

Finally, the implications and potential of this approach to systematic reserve selection are considered.

5.1 Encompassing floristic diversity in reserves

5.1.1 How well does a representative sample of Ecological Vegetation Classes encompass the floristic diversity of the region? - Strategy A

The 30 reserve systems generated by strategy A to include 10% of the area of each vegetation class included records of between 772 and 927 of the region's 1226 indigenous plants (Table 4). This equates to 63%-76% (mean 70%) of the vascular plant taxa in the study. The representation index shows that an average of 13% of the known occurrences of each species were reserved.

For rare or threatened plants, only 61 – 102 of the 187 taxa were recorded within the reserves. This is 33%-55% (mean 43%) of rare or threatened plants in the study.

Table 4 - Occurrence of plant taxa in a representative sample of 10% of the original area of each Ecological Vegetation Class. The results are summary statistics from 30 runs.

	All plants			VROT plants		
	No. of taxa	**% of taxa**	**Representation Index**	**No. of Taxa**	**% of taxa**	**Representation Index**
In region	1226			187		
Mean	857.9	70%	0.133	79.8	43%	0.130
StDev	39.7	3%	0.024	10.4	6%	0.042
Minimum	772	63%	0.084	61	33%	0.065
Maximum	927	76%	0.186	102	55%	0.250

These results show that the representative samples of EVCs were only moderately representative of the region's vascular plants. The absence of records within the reserves for more than a quarter of the taxa in the study, and over half the rare and threatened taxa may be a significant problem if reserves were chosen to be representative of vegetation classes, without consideration of individual taxa. Although the method used precludes a definitive answer, this result does *not* support the proposition that reserving 10% of each vegetation class would create a reserve system representative of a region's floristic diversity.

Note that the method used only accounts for sampled locations of taxa, and does not address the occurrence of these taxa outside sampled locations and times.

Improving representativeness

This result suggests poor representativeness of floristic diversity in a representative sample of vegetation classes. There are a number of possibilities for responding to it. Firstly, one could question the estimate of floristic diversity used, and seek to better represent or sample species distributions. This approach, considered below (section 5.1.3), suggests that a larger proportion of the taxa would have been reserved than was found in the current study.

Secondly, one could seek to increase the total area reserved. While this would be likely to boost the number of taxa reserved, it is often politically and economically difficult to achieve. It also sidesteps the question of how to best locate reserves for a given total reserve area, which is important even if the area available is increased.

Thirdly, one could seek a better surrogate for floristic diversity, so that a sample representative of variation in the surrogate would be more representative of known floristic diversity. A study to directly compare between the current approach, and selection from environmental domains would be useful, and timely. Despite the widespread use of both approaches the author was able to find only one study which compared them directly (Ferrier and Watson, 1997), and none which did so in the context of reserve selection.

A fourth response, and the one trialed in this project, is to ensure that taxa are represented in reserves by deliberately locating reserves to include areas where those taxa have been recorded in quadrats.

5.1.2 Can use of site-based species records in reserve selection increase the floristic representativeness of reserves? – Strategies B, C and D.

The results of strategies B, C and D demonstrate how site records can be used to select areas which include the full diversity of plant taxa in the study. The results show that this can be achieved without increasing the total reserve area, and if required, without compromising representation of any of the vegetation classes. The results, for strategies B, C and D are given in Table 5 below, with strategy A included for comparison. Figure 10 and Figure 11 compare the floristic diversity and representativeness obtained with the different strategies.

Areas selected using quadrat data and EVCs – Strategies B and C

Strategy B sought inclusion of 10% of each EVC and occurrences of all rare or threatened taxa. It resulted in reserves containing an average of 99% of VROT taxa, without an increase in area. The representation index shows that on average just over half the records of these taxa lay within reserves. Although this strategy did not seek them, plants not classified as rare or threatened were quite well represented, with 89% of taxa found in areas selected to meet EVC or VROT targets.

Strategy C sought inclusion of 10% of each EVC and occurrences of all taxa. The resulting reserves included over 99% of plant taxa in the study. Although the target was just 1 occurrence of each species, the mean representation index shows that an average of 42% of occurrences of each taxon and 56% for VROT taxa were reserved.

Areas selected using quadrat data only – Strategy D

With use of quadrat data directly in reserve selection, strategy D selected areas with records of all of the flora of the region and on average, 34% of the planning units where each taxon was found. The result for VROT taxa is similar, with almost 100% recorded in the reserve, and a representation index of 0.48.

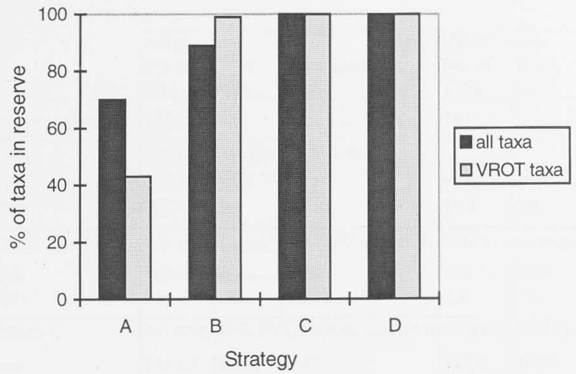


Figure 10 - Mean percentage of taxa reserved by each of the selection strategies. This figure shows how, by using sampled locations for each taxon, areas can be selected to encompass the full floristic diversity of the region. As discussed below in section 5.1.3 however, a quantitative comparison between the result for strategy A and those for strategies B-D must be interpreted carefully, because of the lack of independent validation data in B-D.

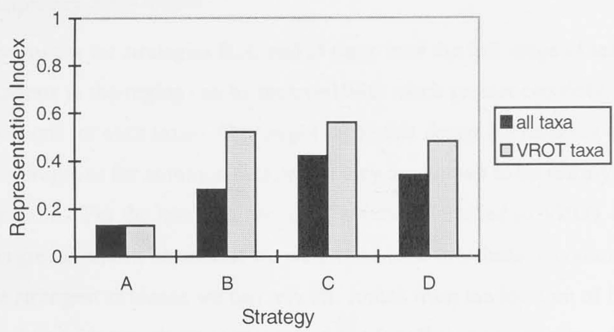


Figure 11 - Representation Index for reserves generated by each of the selection strategies. This shows the mean proportion of planning units with records of each taxon, which were reserved.

Table 5 - Summary of the species occurrence characteristics resulting from the four different selection strategies.

	All plants			VROT plants		
	No. of taxa	% of taxa	Representation Index	No. of Taxa	% of taxa	Representation Index
Total in region	1226			187		
Strategy A	10% EVC target, no area limit					
Mean	857.9	70%	0.13	79.8	43%	0.13
StDev	39.7	3%	0.024	10.4	6%	0.042
Strategy B	All VROT taxa, 10% EVC targets, limited to area required by strategy A.					
Mean	1094.9	89%	0.28	184.3	99%	0.52
StDev	9.6	1%	0.014	2.2	1%	0.017
Strategy C	All taxa, 10% EVC targets, limited to area required by strategy A.					
Mean	1221.2	100%	0.42	185.6	100%	0.56
StDev	1.6	0%	0.006	0.5	0%	0.013
Strategy D	All taxa, no EVC targets, limited to area required by strategy A.					
Mean	1226.0	100%	0.34	186.0	100%	0.48
StDev	0.0	0%	0.005	0.0	0%	0.009

Interpreting these results

The results for strategies B, C and D show how the full range of taxa recorded at quadrats in the region can be reserved with much greater certainty, by selecting known locations for each taxon. One might argue that designing reserves around sample sites is inappropriate for common taxa, when they are known to be readily found at unsampled locations. For the less common taxa, whether classified as VROT or not, this approach has greater merit, because if we want to be sure that these uncommon taxa are reserved, the strongest evidence we can rely on, comes from the location of field survey records. Further, with targets set at the same level for all taxa as was done, common taxa meet their targets so easily that they would have very little influence on reserve selection at all, leaving rare taxa to drive the selection process. This was demonstrated by the result for strategy B, where 89% of taxa made it into reserves despite the fact that the system was only looking for the rare or threatened taxa, which comprise just 15% of taxa.

These results suggest that in practice, selection from vegetation classes or other broad-scale surrogates could be used to provide an acceptable level of representation for the more common taxa, complemented by species-based targets to ensure inclusion of the rarer taxa.[5]

This is the approach used in the Southern Region RFA in NSW, where representation of forest ecosystems is deemed sufficient for most species, but threatened species are considered individually. Where it is not possible to reliably model the distribution of a threatened plant species, representation is based on inclusion of locations known from survey data (Gellie, N., pers. comm. 24/6/99).

The large number of taxa not recorded in the reserves selected for EVCs alone (strategy A) indicates however that the list of taxa requiring special attention would be much broader than a few high profile endangered species. As Figure 3b showed, there are a significant number of taxa which are uncommon in the region, but are not listed as rare or threatened.

5.1.3 Issues in the use of quadrat records in this study

Selection by EVC

Because, in the absence of interpolation, the information on plant taxa is located as points, rather than as a continuous distribution, the results given in Table 4 are heavily dependent on the location of quadrat sites. This occurs in two ways. Firstly, in most cases only a small percentage of the area of a planning unit has been sampled. A single 0.9ha quadrat covers just under 1% of a planning unit. There are thus certain to be additional unrecorded taxa within most planning units. Secondly, a significant proportion of planning units have not been sampled at all. (This relates to planning unit size, which is discussed later in this chapter.)

The consequence of using quadrat data in this way, is that the results generated refer to known or sampled floristic diversity, not actual floristic diversity. They are an under-estimate of actual floristic diversity. It is difficult to gauge the extent to which the number of taxa within a given area is underestimated by using only quadrat records. The analysis of species accumulation curves for each EVC and for the whole region suggests however that a subset of sites is likely to miss taxa within the area it covers. This

[5] A species based approach is less appropriate for life forms which are less sampled, and for which the taxonomy is less well described, such as insects and soil biota. This highlights the importance of including a representative sample of environments in a reserve system.

significantly limits the conclusions which can be drawn about the floristic composition of potential reserves.

It is worth noting however that alternative approaches at the species level are also problematic. Modelled species distributions would solve the problem described above, because they provide spatially continuous information (presence, absence or likelihood of occurrence) and thus avoid patchy distributions which are an artefact of site location. The corollary to this is that modelling techniques introduce a significant element of error. In a test of a range of modelling techniques for flora in Victoria's Central Highlands, using similar flora data to the current project, it was concluded that: 'most species were not well modelled with most of the methods, when judged on the discriminatory ability of predictions for the types of comparisons likely to be required in the RFA process.' (Elith et al., 1998, p.3).

Modelling species distributions was considered as a possible approach for this study, but was rejected for reasons which included a) the need for site records extending beyond the range of the study area; and b) the need for a finer scale digital elevation model (Coops et al., 1998) and accurately geo-referenced quadrat locations (Elith et al., 1998) to be able to include topographic influences on plant distributions in the modelling process.

Selection by taxon

Care is needed in interpreting the results for selection by taxon, because of the lack of independent validation. Whereas strategy A tests the ability of one variable (EVCs) to sample biological variation as represented by another variable (taxa at quadrat sites), this is not the case for strategies B-D. In strategies B-D the measure of species occurrence uses the same data that was used to select the reserves. Lacking independent validation data, the results offer no information about the likelihood of finding those taxa at other locations.

The answer they do provide is about the implications of using taxon records directly in reserve selection. When using quadrat records in this way, the question is not 'can we locate a representative sample of the region's flora?' which is a given with the current method, but rather 'what are the implications of doing so, for the resulting reserve system?' This question points to the results for spatial configuration given in section 5.2.1.

Splitting or subsampling the quadrat data are commonly used to create an independent dataset for validation. These techniques can be used to test a model that predicts presence or likelihood of occurrence of taxa at unsampled locations. These validation techniques were unfortunately not applicable to the current study because, in the absence of modelling, the primary and validation data would rarely overlap. Because few planning units contain more than one quadrat, most planning units selected to include taxa recorded there, would be unsampled by the validation dataset. In other words, the primary and validation data would miss each other, covering different locations.

5.2 Spatial characteristics of the modelled reserves

As shown in Table 6, an average of 841 km^2 or 11.1% of the region was required in strategy A to reserve 10% of each EVC. The other three strategies were capped to the same total area. Strategies B and C had mean areas very close to this threshold.

The spatial characteristics of reserves created using each of the four strategies can be described in terms of two characteristics, or axes of variation:

- spatial variability or irreplaceability; and
- cohesiveness.

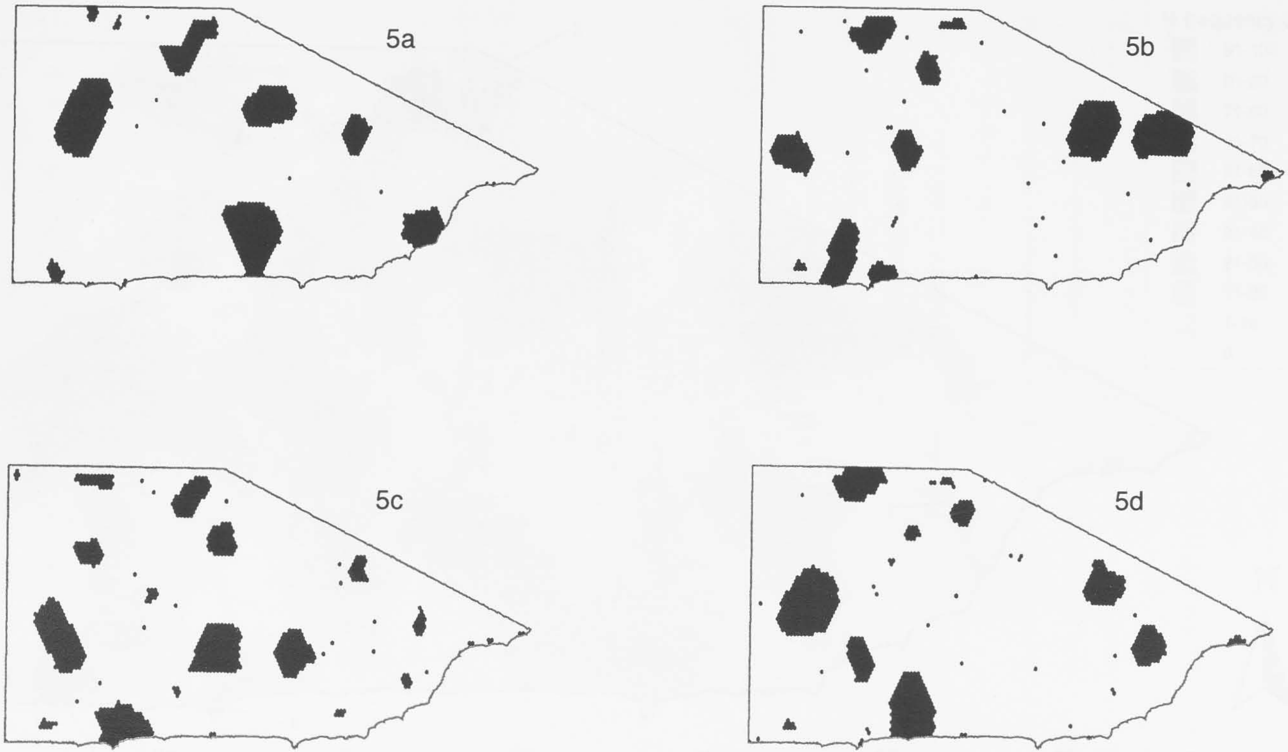
A related factor, the relationship between area and cohesiveness, was examined through calibration of the boundary length multiplier in chapter 4.

5.2.1 Spatial variability

Selection by EVC

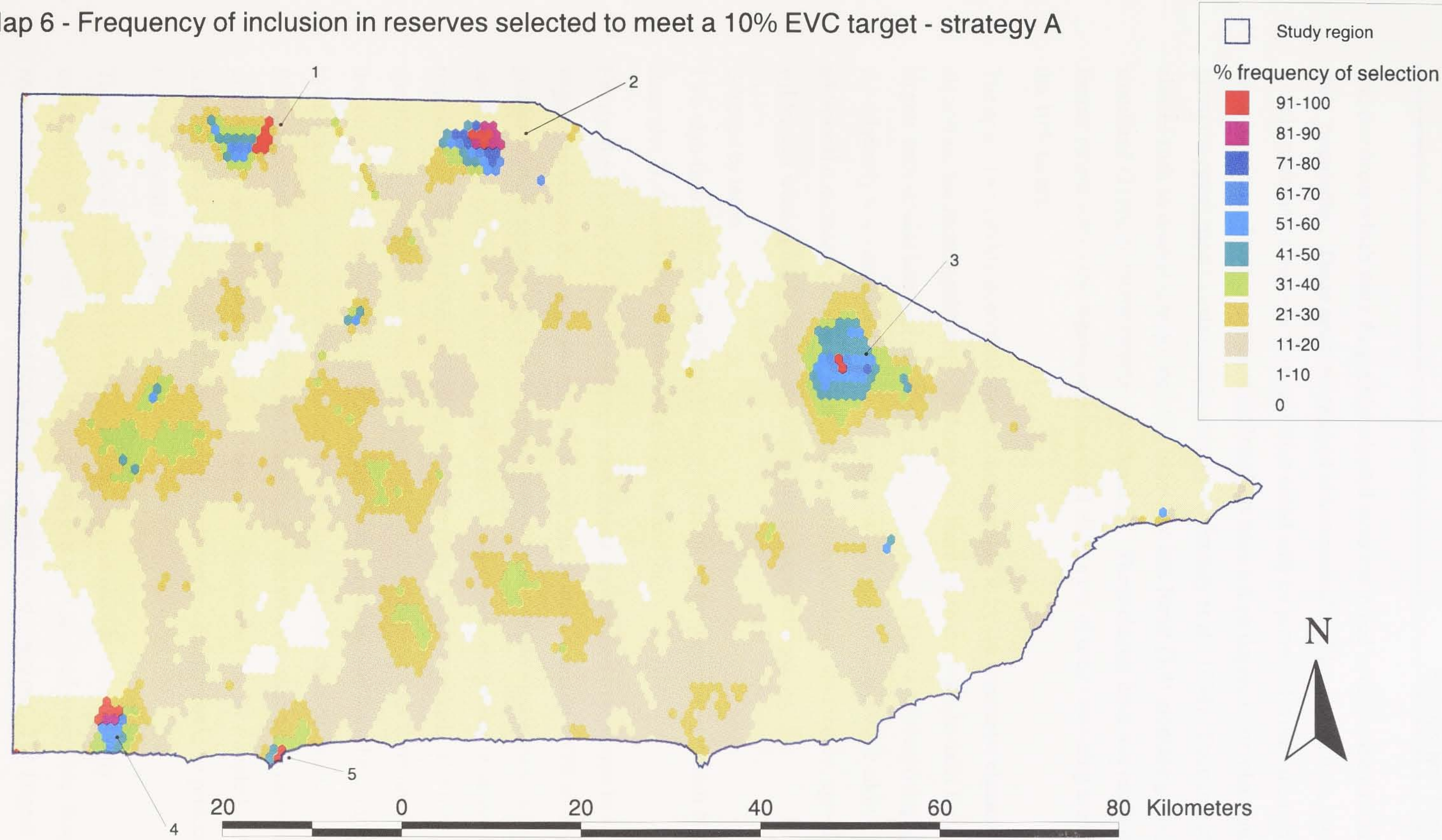
Analysis of the main batch of runs to meet EVC targets (Strategy A) shows a relatively small amount of variation in terms of the total area and boundary length Table 6. The actual shape and location of the reserves is very variable however. A few core areas are selected in all of the outcomes, and a broad biogeographic range is included in each case, but whether and how these areas are connected is very flexible. Maps 5a-d illustrate the variation between reserves designed with the same parameters, under strategy A. Map 6 shows the frequency with which different areas were selected. It highlights the great flexibility in choosing areas which meet the representation targets. Although the areas selected in each run occupy about 11% of the region, 88% of the region was selected in at least one of the runs. The few areas selected more than 9 times in 10 come to less than 3% of the average area reserved. Areas selected more than half the time come to just 10% of the average area reserved.

Maps 5a-d - Variation between reserves generated with the same parameters and targets



Maps 5a-d show reserve options which were each produced using strategy A to meet the 10% EVC target with a BLM of 1. The variation between the different runs illustrates two points. Firstly, the way that a stochastic algorithm generates different outcomes for the same input data and settings. Secondly, it shows that there is great flexibility in which areas may be used to meet the EVC targets.

Map 6 - Frequency of inclusion in reserves selected to meet a 10% EVC target - strategy A



Map 6 illustrates the high degree of variability in the areas selected to represent 10% of each Ecological Vegetation Class. Very few areas were required in all cases, while most areas were chosen at least occasionally. This indicates a high degree of flexibility in meeting the targets under strategy A. The few areas which were irreplaceable under this strategy are identified with pointers and discussed in section 5.2.1.

The few areas which were frequently selected were examined to determine the reason for their 'popularity'. These areas are marked and numbered on Map 6. All were found to contain uncommon vegetation types which could only be protected within the region at those locations. In other words, those locations have a high degree of irreplaceability, given the reservation targets used in strategy A (Pressey et al., 1994). They include classes such as coastal grassy forest (5), limestone box forest (3,4) montane grassy woodland (2) and montane riparian woodland (2). These classes have lost much of their former extent within the region, and thus need all or most of their remaining area to meet the 10% target.

The spatial variability described above is made possible by the stochastic element in the algorithm, but more significantly, it shows that there are many ways to meet the EVC targets with similar levels of efficiency and cohesiveness. This is not surprising, given the relatively few constraints in this strategy. Raising the EVC targets, or adding additional constraints, related to plant taxa, or to fauna habitat, land tenure, age-class, or wilderness would greatly narrow the variability in areas selected.

Selection by taxon

Two significant effects are evident from analysis of the frequency of selection of areas in strategies B, C and D, as shown in Maps 7a, 8a and 9b.

Firstly, there is far less variation in the areas selected, compared to selection by vegetation classes discussed above. The proportion of areas selected >90% of the time is greater than for strategy A, as is the proportion of areas selection more than half the time and the proportion of areas not selected at all. These effects are greater in strategy C, requiring all 1226 taxa, than in strategy B where only 187 are required. The more specific and numerous the goals for representation, the fewer options there are for meeting those goals. The degree of irreplaceability increases, and thus the compromises to be made between conservation and other land uses become more difficult.

Secondly, the quadrat sites plotted on Maps 7a, 8a and 9b show that while there was a preference in selection towards sampled areas, in many cases heavily sampled areas were selected rarely or not at all, indicating that the taxa recorded there could be protected more efficiently elsewhere.

This is clearest in strategy D (Map 9b) where quadrat records were the only environmental data used. Among the locations that were selected every time, three larger areas which were irreplaceable for the taxa recorded there, stand out. The largest of these is in the catchments of Goongerah and Stony Creeks just south of Bonang (1), where the terrain drops steeply from the Errinundra Plateau. The other areas are in the Yalmy catchment, east of the Snowy River (2) and on the east side of the lower Genoa River (3). These areas could benefit from further study to determine the reasons for their consistent selection, and hence whether they should be identified as places of particular significance for plant conservation. It is possible that areas (2) and (3) which are close to the edge of the study region, may represent outliers of conditions and taxa more common outside the study area.

5.2.2 Cohesiveness

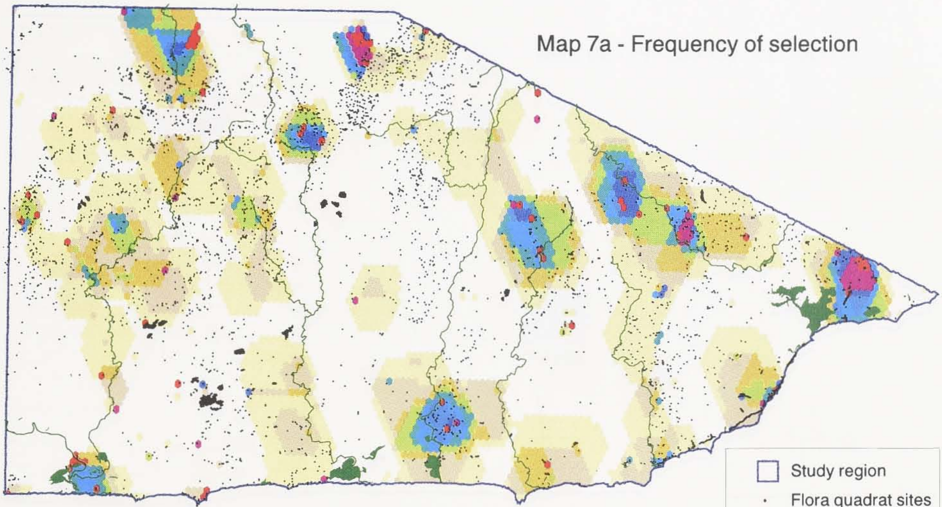
Although use of species data was able to include records of all taxa without enlarging the reserve system, this occurred at a large cost to reserve cohesiveness. As shown in Maps 7b, 8b, and 9b, the inclusion of species targets in strategies B-D significantly decreased the cohesion of the reserve system compared to strategy A. The increase in the perimeter-area ratio from strategy A, was substantial for strategies B (22%) and C (51%), but greatest for D (393%) which had no EVC targets. This effect is shown in Figure 12.

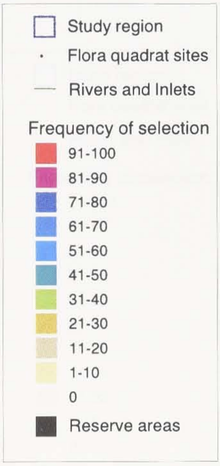
Strategies B and C are most readily comparable to A, in that they each met the same EVC targets and occupied close to 11% of the region. Moving from the EVC targets in strategy A, to add targets for VROT taxa (B) and then targets for all other taxa (C) the reserve areas become progressively smaller and more numerous, and the number of single, isolated planning units selected increases. By seeking to represent biodiversity as seen through the fine filter of taxonomic diversity, the number of conservation values to be considered greatly increases, many more of them area rare, and the likelihood of areas being described as equivalent in terms of the biodiversity they contain decreases. Thus there are more places which are required, and less options to create contiguous reserve areas.

The fragmentation effect was most extreme for strategy D however, which was the only one which did not require a minimum area of each EVC. It used only 3.2% of the region to represent all taxa. Although strategy D selected areas with records of all taxa in the study it did so, as Map 9a illustrates, with minimal cohesion. This is because without any area based targets, it selected the required species within the minimum area, which in many cases meant just the planning unit in which a particular quadrat site occurred. While selection using site records alone provides a useful indication of a minimum set of

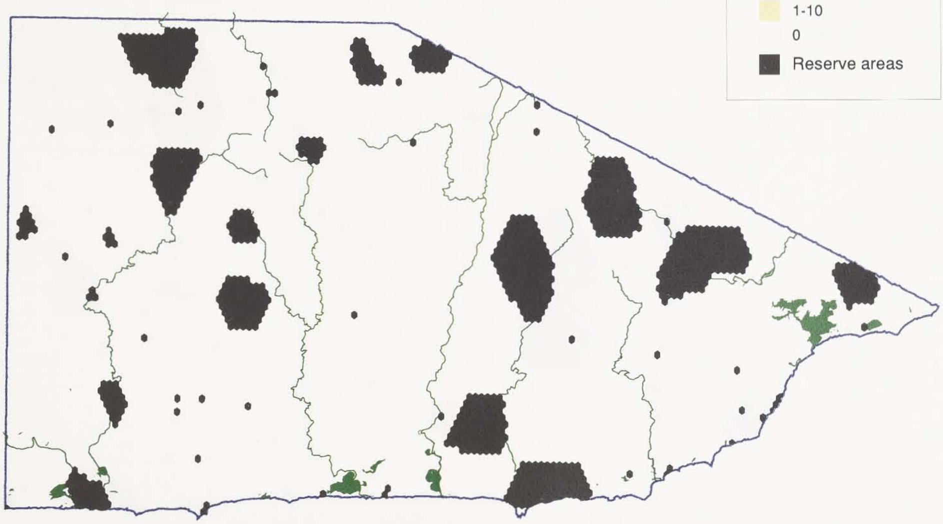
Strategy B - 10% EVC target, at least 1 sample of all rare or threatened taxa, limited area

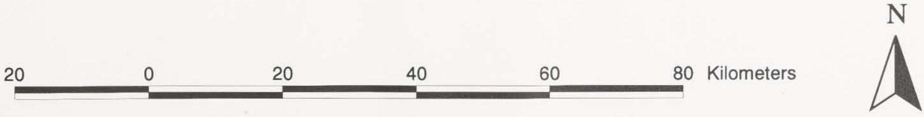
Map 7a - Frequency of selection



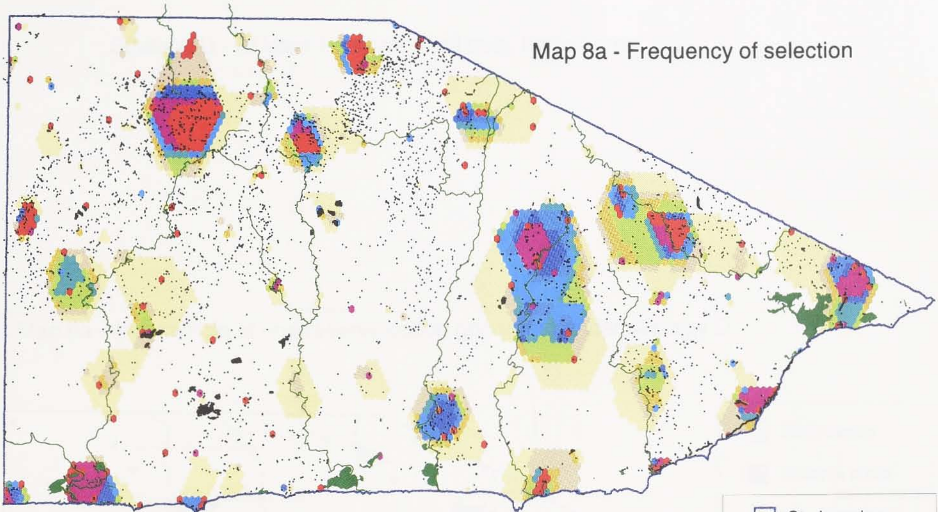


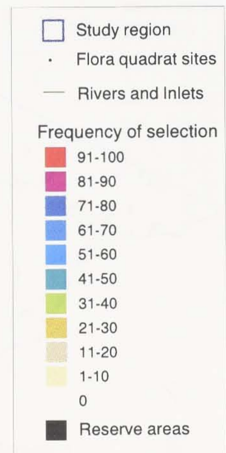
Map 7b - An example of one reserve option generated under strategy B



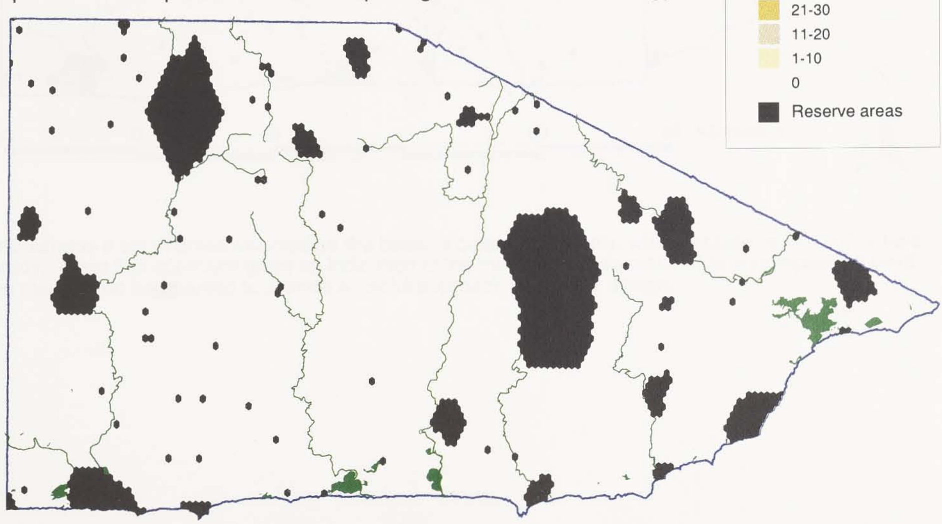


Strategy C - 10% EVC target, at least 1 sample of all taxa, limited area



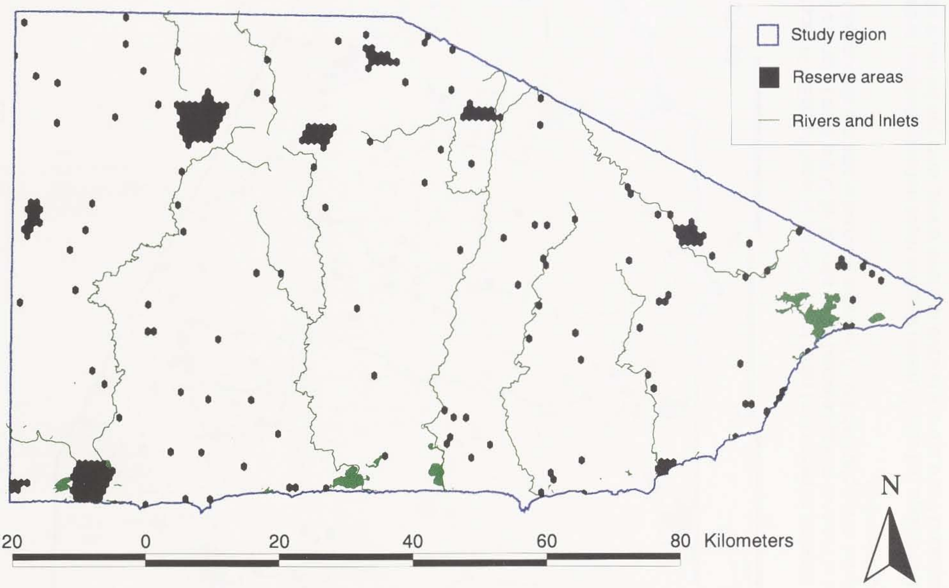


Map 8b - An example of one reserve option generated under strategy C



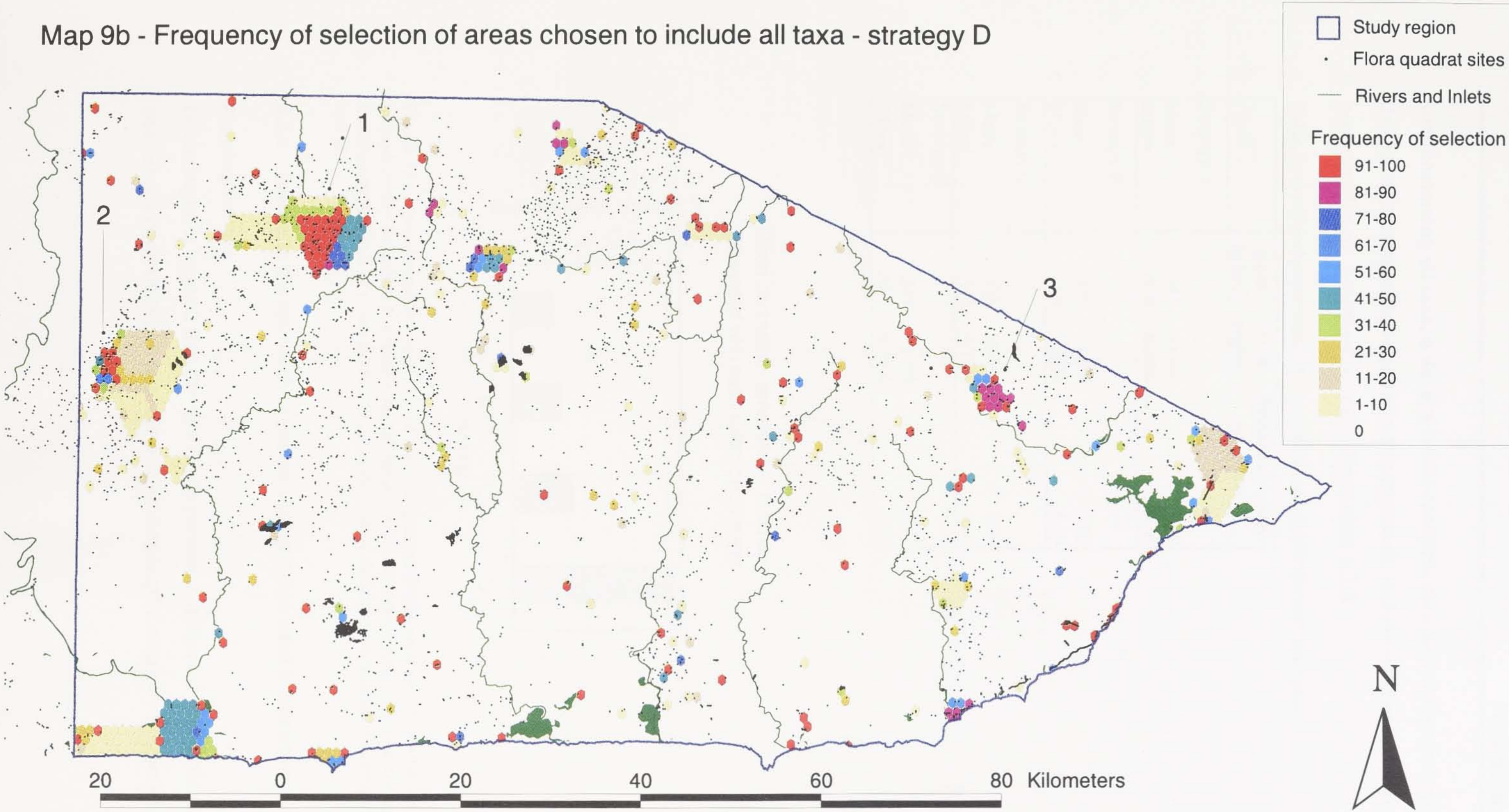
Strategy D - At least 1 sample of all taxa, limited area

Map 9a - An example of one reserve option generated under strategy D



Map 9a illustrates a set of areas selected on the basis of quadrat data only, which include records of all taxa in the study. While this approach gives an indication of the minimum areas required to encompass all plant taxa, the result is too fragmented to provide a useful approach to reserve design.

Map 9b - Frequency of selection of areas chosen to include all taxa - strategy D



Map 9 shows that strategy D, driven largely by the location of records of rarer taxa, offers the least flexibility in locating reserves. This is apparent by comparing this map to Map 6, where the proportion of reserve areas selected all or almost all of the time is far lower. The finer grain representation of biodiversity at the taxon level results in a higher degree of irreplaceability. The three marked areas are discussed in section 5.2.1 as possible areas of endemism which may be of significance for plant conservation.

Although selection is biased by quadrat distribution, this does not simply mean that the most densely sampled areas will be selected. Many well surveyed areas, were not chosen, because the taxa found there are not unique to those areas, and can be more efficiently reserved elsewhere.

sites containing all taxa, it did not in this case provide a useful approach to reserve design. Potential improvements to this approach, including better calibration of the boundary length multiplier are raised in section 5.2.3.

Table 6 – Spatial characteristics of reserves generated with the four strategies.

	Area in km^2	% of region	Perimeter – area ratio (km^{-1})
Strategy A			
Mean	841	11.1%	0.59
StDev	28.9	0.38%	0.06
Strategy B			
Mean	829	11.0%	0.72
StDev	15.0	0.20%	0.04
Strategy C			
Mean	823	10.9%	0.89
StDev	9.3	0.12%	0.03
Strategy D			
Mean	245	3.2%	2.32
StDev	34.5	0.46%	0.06

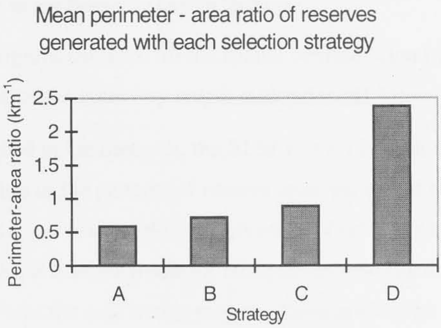


Figure 12 - Mean perimeter - area ratio of reserves generated with each selection strategy. Representation of biodiversity described at a finer level can be achieved without expanding the reserve system, but there is a corresponding loss of reserve cohesion, as indicated by the higher perimeter-area ratios for strategies B,C and D.

5.2.3 Issues in the approach to spatial configuration used in this project

Planning unit size

In hindsight, it appears that the size of the planning units used was problematic. Whereas smaller planning units can produce a more meaningful result by responding to environmental variation at a finer scale, they also run the risk of dividing the landscape at a finer scale than is justified by the data used.

The planning units in this study were originally planned to match grid cells in the digital elevation model (side length 0.003 degrees) which was to define the scale of modelling for the study. With the decision not to model, the shape was changed to hexagonal, and the area increased from approximately 8ha to 92ha as described in the method.

While this size was appropriate to the scale of the ecological vegetation class mapping, it appears that for the point-based flora quadrat data, a far larger planning unit size would have been more appropriate. The size was sufficient to allow for the estimated positional error of quadrat locations, but should have been far larger to allow for variations in sampling density, and to reduce the proportion of unsampled planning units.

In comparison, other published studies which used raw (unmodelled) site records had larger planning units. Lombard et al. (1997) used 900 ha units, while Kirkpatrick and Brown (1994) used a 10 000 ha grid fitted over the shape of Tasmania. The 900ha size would probably be in the right order of magnitude for the data in the current study.

Calibration of the Boundary Length Multiplier

Another significant issue for the spatial configuration function in Spexan was the calibration of the boundary length multiplier (BLM).

As described in the methods, the BLM was set in each case primarily from visual examination of the patterns of reserve areas generated at a range of BLM settings. While this approach is reasonable for a given set of environmental data (eg the EVC data in strategy A), it is problematic for comparison between data expressed at different spatial scales, as was the case in this project. Compared to the continuous distribution of the EVC data, the quadrat data was expressed at a different spatial scale, producing more scattered distributions and thus longer boundary lengths.

A difficulty in adjusting the BLM to compensate for this factor to allow a direct, quantitative comparison between strategies A – D, is that the difference derives from two distinct causes. One is the difference in spatial scale of the two datasets. Scattered, discontinuous quadrat locations favour scattered, discontinuous reserves, unless a very high BLM is used to overcome this tendency. This is a function of the data used.

The second cause is the difference in the level of description of biodiversity. The recognition of 1226 taxa, many of them rare, means that a wider and more scattered range of areas would be required to include all taxa. This is a reflection of the environmental and taxonomic diversity of the region, not the method.

One approach suggested for calibrating the BLM across data types would be to adjust it so as to hold the perimeter area ratio constant across all strategies. The species richness achieved with a constant area and degree of cohesion could then be compared. In seeking to compensate for the difference in spatial scales however, this approach could mask out genuine differences in dispersion which result from a finer level description of biodiversity.

These problems prevented a quantitative comparison between the strategies, but could not obscure the overwhelming effect, that representation of all taxa required a far more dispersed reserve network than representation of all vegetation classes. This effect was apparent even when the BLM was increased by a factor of 20 to compensate for the point based quadrat data.

5.2.4 Resolving the problem of scale

Scale is a common theme among the methodological problems discussed above. While most elements of the study worked well, it was the difference in scale between continuous and point data which caused problems in several areas of the study, and prevented more rigorous quantitative comparison between the four selection strategies.

The general lesson to be learnt from this, is of the importance of comparing like with like. Surrogates may be evaluated by comparing sampled points with the value of the surrogates at points (Ferrier and Watson, 1997). Introducing spatial factors such as area and adjacency however requires that data be in a consistent format if valid comparisons are to be made.

Although not without its own problems of accuracy, the use of modelled species distributions or other continuous data could address virtually all of the particular methodological weaknesses described for the current study. These include:

- Absences by default at all unsampled locations, which would be covered by modelled presence, absence or probability estimates;

- Lack of validation: data splitting or repeated subsampling could be used to assess taxonomic richness of potential reserves against data independent to that used for reserve selection;
- Different spatial scales of data layers: continuity of information in all data layers could remove the main barrier to consistent calibration of the BLM discussed above.

However, modelling would be unlikely to be useful for taxa known from a very small number of records. This fact, and the sheer amount of data involved in models for a large number of taxa would likely prevent such a study from including the full range of sampled taxa, as was done in the current study.

5.2.5 How do these results apply to real-world conservation strategies?

A factor common to all four strategies (to differing extents) is that to meet their representation targets all include small isolated reserve areas. As discussed in chapter 2, such areas may in many cases be too small to support the complement of species currently occurring there, in the long term. Further, it would be quite impractical to manage so many small areas as separate conservation reserves. Leaving them out however would result in a reserve system which is not fully representative of its rarest elements.

There are two different ways of interpreting this. One interpretation is that the results simply demonstrate that despite meeting the IUCN goal of 10% representation, the area allocated to reserves in this study is not sufficient to represent the broad range of identified attributes within a realistic configuration of reserves. From this viewpoint, one would need to increase the total reserve area until sufficiently cohesive reserves can be produced. Alternatively, one might need to accept that the biological variation in the region can't simply be crammed into 10-12% of the land no matter how thorough the selection process, and then make some hard decisions about which taxa or vegetation types should have the priority for reservation.

Although such a finding may be important to temper expectations that adequate conservation can be achieved within small areas, it should come as no surprise. As Ralph Slatyer observed albeit at a larger scale:

> Regardless of the size of a reserve, it must be realised that it is impossible to maintain as great a degree of species diversity in reserves as on a whole continent. This is partly because it is not possible to include every aspect of continental habitat diversity, every ecological niche, in reserves; and partly because in any assemblage of species from a finite area of land, some species are found to be rare, while others are found to be abundant. (Slatyer, 1975, p.24)

An alternative interpretation of these results is as a guide for integrated management of the region. In this approach the distinctions between reserved and non-reserved areas would be less sharp. Considered within a matrix management approach, the large and smaller reserves are just areas of special management within a broader landscape in which the surrounding matrix is integral to the conservation strategy. In this approach, the small reserves could in effect be zones managed for the benefit of particular taxa or communities, within a patchwork of areas all managed for particular values, and for different degrees of production and conservation depending on their particular characteristics.

This accords with a growing body of thought in conservation science and policy which argues that long term biodiversity conservation cannot be achieved by relying solely on reserves (Kanowski et al., 1999). Managing the forests outside of reserves sympathetically to buffer, link and complement the protected areas can transform protected areas 'from isolated islands to integrated networks of functional ecosystems.' (Kanowski et al., 1999, p.39). If such a strategy could be applied, then reserves such as generated in this study might form the nucleus of an effective conservation strategy for the region.

The effectiveness of such an approach however would depend on a number of factors, firstly, the particular needs of the taxa and communities in question. Some may thrive on disturbance, or do fine in small patches, especially if linked by similar (although more heavily disturbed) vegetation. For other taxa however, where the edaphic or biotic effects of fragmentation reduce the viability of a small remaining population, such an approach may produce poor conservation outcomes. Further, such an approach would rely on both a high level of understanding of the management requirements of a diverse range of biota, and the resources and willingness to carry out such management. With the resources available to both nature conservation and forest management agencies stretched as they currently are, the degree of research, monitoring and careful management required to responsibly implement such an approach would be difficult to achieve.

The extent to which reserve systems proposed by this study could be effective for long term conservation, would depend to a significant extent on how well the surrounding matrix can be sympathetically managed, to buffer, complement and link the populations in small protected areas.

5.3 *Outcomes of the case study*

The outcomes of the case study can be summarised as three broad conclusions.

- Firstly, a representative sample of 10% of the estimated pre-European area of each ecological vegetation class within the region does not constitute a good representative sample of the region's floristic diversity as known from quadrat records. Less than three quarters of plant taxa and less than half of the rare and threatened plants of the region were generally recorded in the modelled reserves. Although the method used under-represents species occurrence, this result suggests that the proposition that a representative sample of vegetation classes represents the biodiversity of a region, should be treated with caution.
- Secondly, areas can be selected to represent the full sampled floristic diversity of the region within the same total area by locating reserves to include recorded occurrences of particular taxa.
- Thirdly, this increase in recorded floristic diversity occurred at a great cost to the cohesiveness of the reserve system. Driven largely by the requirement to include a large number of rarely recorded taxa, reserves selected to maximise floristic diversity within a limited area were too fragmented to be effective for long term conservation, within a traditional approach to reserve design. The configurations produced could potentially be more relevant to a matrix management approach in which the non-reserved areas are managed sympathetically to link and complement the populations occurring within protected areas.

5.4 *Broader issues in method for systematic reserve design*

This section goes beyond the particular results of the case study, to briefly consider some more general issues for systematic selection of reserves.

5.4.1 Rarity

Rarity has been approached in two ways in this study. The Victorian Rare or Threatened plant classification has been used to analyse the reservation status of these plants as a separate group, and to set conservation goals (in strategy B). The number of records of each taxon also functioned as an implicit measure of rarity in the selection algorithm.

A more sophisticated view of rarity however would need to consider different types of rarity and hence different reasons for taxa occurring at low frequency in survey records.

For example, a species may be abundant over a small geographic range, or locally common at a small number of disjunct locations where their narrow habitat requirements are met. Other species are distributed across a wide geographic range, occurring sparsely through a broad range of habitats. The variety of forms of rarity can be defined in terms of three characteristics, geographic range, habitat specificity and local population size (Rabinowitz, 1981). To this a fourth characteristic may be added, which is habitat occupancy, the tendency of a species to occupy a larger or smaller fraction of its potential suitable habitats (Rey Benayas et al., 1999).

Different types of rarity imply different conservation approaches. Use of site records to locate reserves for a highly localised species (assuming it needs protection) would be more appropriate than doing so for a widely scattered species which is likely to occur (albeit at low frequency) at many unrecorded locations. For the localised species, quadrat records tell us something important about where it lives, whereas the location of records for a rare but widespread species may be more a matter of chance. The utility of using modelling techniques to define areas of suitable habitat may depend on the consistency with which the species in question occupies that habitat.

The sampling process itself is also a factor in the apparent rarity of some taxa. Cryptic species may be overlooked, mis-identified as a related species, or described only to the genus level. Choice of site location may tend to under-represent taxa which occur in particular environments, such as ecotones (Elith et al., 1998) or rugged terrain. Taxonomic uncertainty and taxonomic change are also significant issues. There is, for example, substantial difference in the taxa described for the study area in the two flora guides used in this project (Willis, 1972; Entwisle et al., 1993). The changes relate to both increasing knowledge of the flora, and also changing fashions about the appropriate level of taxonomic division or aggregation. Reclassification of old records to reflect a new taxonomic structure may be difficult or impossible, leading to the apparent rarity of taxa which have few records because they are either newly or no longer recognised and recorded in the field.

The use of records for individual species in reserve planning should thus be undertaken with care, and ideally with careful consideration of how the factors described above apply to each taxon.

5.4.2 Data requirements

As discussed at various points in this paper, issues of data type, scale and distribution are the key to the types of analysis which can be done, and the degree of confidence which can be placed in the results. Conducting targeted surveys to fill sampling gaps in environmental space, such as areas identified in Map 3, and to fill larger sampling gaps in geographic space would be an important (but expensive) step in this direction.

As discussed above, environmental modelling to interpolate species distributions between sampled points could provide a better estimate of species occurring in a given area, and also enable better handling of spatial configuration issues by providing environmental data at a consistent spatial scale. Environmental modelling would benefit from strong links between the flora survey data and environmental variables for the same locations. Three steps which would help to achieve this are:

- recording of matched biotic and abiotic data at survey sites, to cover vegetation structure and floristics as well as environmental measurements such as slope, aspect, elevation, soil type and depth;
- accurate recording of site locations to within a few metres using differential GPS technology;
- availability of a finer scale digital elevation model, preferably at a 25m resolution.

Information in this format would enable models to take account of a wider range of factors correlated to species distribution, including important topographic influences on species distribution, such as shading and soil wetness.

5.5 Lessons learnt from the use of Spexan

Transparency

One issue in the use of simulated annealing, along with linear programming, is that of transparency. Because many factors are considered concurrently, it is hard to know just why a particular area was selected. One can go back to the source data to look for significant features at that location, however in many cases the selection of that area will result from a combination of many factors. Further, unlike iterative heuristic selection methods, the recommended areas are not given an order of priority to assist with decision making. A solution is presented as a whole, with no cues as to the relative importance of different areas. One option for getting around this would be to order the areas selected in terms of their irreplaceability.

Optimality

The fact that the algorithm cannot identify the globally optimal solution, but instead identifies a range of locally optimal solutions, may at first appear as a weakness in the approach. I would argue however, that it may actually be an advantage. Rather than producing a single 'black box' reserve design, this approach recognises that in most cases there are a range of solutions which will meet the specified criteria almost as well, but could be more acceptable for real-world implementation. Looking at the frequency of selection, as was done in this project, enables differentiation of irreplaceable areas, which should be non-negotiable, from those areas which could be easily substituted for others.

Complexity

Spexan deals well with the simultaneous consideration of a large number of values, which is the basis of the reserve selection problem. It has no problem dealing with factors such as a change in the availability of areas for reservation, or in the priority placed on particular values. Through this project, Spexan and its simulated annealing algorithm demonstrated an ability to handle very complex reserve design problems involving over 1200 conservation values distributed across more than 8000 planning units. This is important, given the complexity of real world conservation questions. The complexity of the problem was limited in this case by the necessity for the user, rather than the computer, to have a simpler problem. For example, it would have been easy to include individual representation targets appropriate to each plant species. The real challenge would be to determine such targets. The version used in this study in fact contained additional features for use in other projects, such as the ability to specify a minimum patch size and maximum separation distance of suitable habitat for each species, so that areas not likely to provide viable habitat would not be counted towards the habitat area required for that species. The point is, that technical advances of this type are of value so long as they function to extend the application of ecological knowledge, rather than substitute for the lack of it.

5.6 Opportunities for further work

This study could be seen as an initial exploratory stage of a larger investigation. A number of avenues for further study present themselves. Firstly, repeating the analysis performed in this study, but using modelled species distributions could provide a more definite answers to the questions posed in this thesis, namely the effectiveness of using vegetation classes as a surrogate to sample floristic diversity, and the implications of selecting for individual taxa instead. It may be necessary to work with a subset of taxa. A rigorous comparison between selecting by vegetation class and selecting by taxon might be achieved by comparing the floristic diversity of the reserves generated, while holding the area and perimeter - area ratio constant.

Secondly, the reserve selection techniques trialed and compared in this project could be made more robust through a systematic analysis of the sensitivity and calibration of the different parameters used across different study types and regions. This could include the relationship between total area and representativeness, and more rigorous methods of calibrating the spatial component of the model.

Thirdly, it would be useful to go beyond the general finding that a sample of ecological vegetation classes missed records of many species, to examine the reasons behind this. One approach to this would be a numerical classification using PATN to analyse the variation in floristic composition both between and within vegetation classes, to see where and how well the classes match patterns of floristic variation.

A comparison of the effectiveness of selecting reserves using EVCs and environmental domains, would be useful, as a follow-up to the work of Ferrier and Watson (1997). It could easily be performed using the data as set up for the current project. Such a comparison could use either the reserve selection approach established in the current project, or a comparison of species accumulation curves generated by selecting sites either within or across surrogate classes.

Ultimately however, if the need to consider reserves in the context of managing the whole landscape is taken seriously, then systematic planning tools will need to consider more than just reserve selection. Research could usefully be directed to algorithms which go beyond binary selection (reserved or not reserved) to rationally allocate areas to a number of management zones. It remains to be seen to what extent automated systems would provide useful assistance for such complex problems. They would be challenged by both the complexity of the factors and trade-offs involved, and by the fact that the resulting advice may be difficult to integrate into processes by which land use decisions are made, which are social and political as well as technical.

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