# A review of biodiversity investment prioritization tools

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**Executive Summary**

There is a need for increased transparency, accountability and efficiency for public investment in biodiversity conservation and restoration. Prioritization and decision tools may assist natural resource managers achieve systematic and defensible biodiversity investment decisions. However, there is a confusing array of tools, resources, and decision frameworks available to managers and decision makers. Here I review some of the more commonly used tools, identify where the various tools and resources fit into an overall biodiversity investment prioritization strategy, and provide a brief evaluation of each of the tools and resources reviewed. Not every tool used in conservation planning in Australia is reviewed here, though a range of tools are presented that cover the general steps in conservation investment planning. I have done my best to mention the tools that are not reviewed in detail in the appropriate sections. I conclude that there exists a large array of tools to support biodiversity investment prioritization. However, there is a need for a careful examination and trialling of the existing tools in an NRM biodiversity investment prioritization context to clarify which of the tools are most appropriate for resolving NRM prioritization issues. There is little guidance on how to reach consensus about the values that underpin conservation investment and appropriate management goals. There is a need for further research and investment in social scientific methods for eliciting values and determining and setting goals for biodiversity conservation. Existing tools give limited consideration to public versus private benefits and strategies for ensuring successful adoption and implementation. Very little attention has been given to methods for estimating the probability and magnitude of threat reduction that can be achieved under different management options, though some of the tools described could be adapted to do so. Table 1 provides an overall summary of the tools reviewed here. Each of the tools mentioned in Table 1 is reviewed in more detail in body of the report. All of the tools described in this report have the potential to be misapplied or applied beyond their intended use, resulting in spurious or misleading results. Care must be taken in utilizing any of the methods to become familiar with their original intended use, their assumptions and their limitations.

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| Tools and Frameworks (section) | Primary purpose | Technical demands (H,M,L) | Subjective (H,M,L) | Widely used (Y/N) | Spatial data necessary (Y/N) |
|---|---|---|---|---|---|
| Biodiversity Action Planning (1.1) | A planning framework | L | H | Y | N |
| Conservation action planning (1.2) | A planning framework | L | H | Y | N |
| Actions for Biodiv Conservation (2.1) | Monitoring and data storage | M | M | N | N |
| Vegetation mapping (2.2) | Measure/Map assets | L | M | Y | Y |
| Veg Condition mapping (2.3) | Measure/Map assets | L | M | Y | Y |
| Habitat modelling (2.4) | Measure/Map assets | H | M | Y | Y |
| Biodiv planning assessment (2.5) | Measure/Map assets | M | L | N | Y |
| Marxan with zones (3.1) | Identify priority zones | H | L | Y | Y |
| C-plan (3.2) | Identify priority zones | H | M | N | Y |
| Zonation (3.3) | Identify priority zones | H | L | N | Y |
| Bioregional network analysis (3.4) | Identify assets/action | L | H | Y | N |
| Optimal restoration altered habitat (3.5) | Identify priority zones | H | L | N | Y |
| Project prioritization protocol (4.1) | Measure and rank actions | L | M | N | N |
| Biodiversity benefits index (4.2) | Measure and rank actions | L | H | Y | N |
| BioRisk (4.3) | Measure and rank actions | M | M | N | N |
| Population modelling (4.4) | Measure and rank actions | H | M | N | N |
| Bio-Forecasting (4.5) | Measure and rank actions | H | M | N | Y |
| Strategic landscape investment (4.6) | Measure and rank options | H | M | N | Y |
| BioPrEP | Measure and rank options | M | M | N | Y |

**Table 1.** A summary of the attributes of each tool/framework reviewed. The technical demands column is based primarily on the need for expert ecological, modelling or computer programming skills and knowledge. If technical demands are noted as low, it doesn't mean that the availability of ecological/biological or modelling expertise wont add to the utility of results obtained, it simply means that *a* result can be obtained without those specific skills. The "subjectivity" ranking largely depends on the extent to which the methods requires documentation of all assumptions and whether the method primarily relies on measured attributes or peoples opinion/gut-feeling. Generally, methods that are more subjective are less transparent and repeatable. The "spatial data necessary" column refers only to whether spatial data are required to utilise that particular tool. In many cases, spatial data may be 'subjectively' derived expert data that is converted to a map using any of a number of interpolation methods.

## Introduction

There is a growing recognition that investments in biodiversity conservation and enhancement should be strategic and produce the greatest possible benefit for the money and resources invested (Possingham 2001). The public and governments at all levels now expect biodiversity investments to be *demonstrably* efficient; this is central to government's credibility as a prudent investor (ANAO 2008). The need for transparency and accountability in natural resource investment and the ecological and institutional complexity of biodiversity management reinforces the importance of systematic decision protocols (Duncan & Wintle 2008) because ad-hoc decision making often lacks accountability, transparency, and efficiency.

There exists some confusion among NRM (and many other) organizations about the existing resources and best approaches for achieving defensible and efficient decisions about investment in biodiversity. There are a confusing array of computer programs, data bases, decision support tools, rules-of-thumb, frameworks and philosophies documented in the expansive scientific literature on conservation planning and biodiversity management. A *Web of Science* search for "conservation planning" returns more than 1000 papers published in peer reviewed journals in just the past five years. The aim of this review is to provide a summary of the more widely used resources, tools and frameworks. An exhaustive review of all available resources is beyond the scope of this paper, though I have, where possible, tried to reference tools that I am aware of but don't review here in detail.

*What do NRM groups need in terms of biodiversity prioritization and decision tools?*
There is a tension in biodiversity management between wanting to represent and incorporate ecological processes and complexity and needing to maintain practicality and expediency due to limited time, resources and knowledge. Put another way, it is important that investment in biodiversity is based on rigorous science, agreed conservation planning principles and the best available knowledge, but equally important that time is not wasted and that methods used to prioritize investment are simple and practical enough that they can be understood and used by the people responsible for the decision process. Here I evaluate the range of resources, tools and decision frameworks in terms of (i) rigour and adherence to conservation planning principles, and (ii) how accessible and difficult they are to use. I structure my inventory of biodiversity prioritization tools according to where they fit into three steps of biodiversity investment prioritization:

(i) identifying values, assets, goals, and threats;

(ii) identifying priority areas and a set of potential management actions; and

(iii) ranking/prioritizing and implementing those actions within the available budget.

As well as reviewing the tools available for each of these steps, I review some conservation frameworks that incorporate some or all of these steps. Some of the frameworks reviewed break these three key elements down into a greater number of steps. 'Adaptive' frameworks, such as The Nature Conservancy's *Conservation Action Planning* (CAP) framework (see also Radford et al. 2007, Duncan & Wintle 2008) include an extra step of monitoring and evaluation. There is almost no guidance or support for monitoring and evaluation of biodiversity investments at any level of jurisdiction (including the CMA/NRM group level) which has resulted in an absence of strategies for adaptive management of biodiversity. The issue of monitoring and evaluation of investments is beyond the scope of this review and is discussed only in passing.

## Frameworks and tools for biodiversity

### 1. Overview of some existing *frameworks*

There are some frameworks for biodiversity management that are relevant to this review. I define a framework as something that attempts to guide practitioners through the various steps of biodiversity management, which sometime includes a prioritization step and sometimes does not. Most agencies and organizations have some kind of framework for biodiversity prioritization and management. Here I focus on just a few emblematic examples. There are also some frameworks identified in the academic literature that attempt to order the steps in biodiversity management. Margules and Pressey (2000) provide an overview of general conservation planning principles. Lindenmayer et al. (2008) provide a checklist of considerations for conservation management of landscapes and detailed discussion and case-studies are provided in Lindenmayer & Hobbs (2008). Radford et al. (2007) provide a good overview of the principles behind landscape restoration for biodiversity. In that paper they provide some 'rules-of-thumb' that loosely follow adaptive management principles in defining and assessing the landscape, developing a vision and objectives, identifying potential actions and measures of success, implementing actions and monitoring the results to provide feedback on best approaches to future management. A similar framework is promoted by Duncan & Wintle (2008) for iteratively reducing uncertainty about best vegetation management options using formal model-based adaptive management. Finally, indicator species concepts, including the focal species concept (Lambeck 1997) provides a framework for scaling down the number of biodiversity elements that need to be considered in conservation prioritization and planning. These concepts have been criticized due to an apparent lack of generality and because knowledge about taxon-based indicators is limited (Lindenmayer et al. 2000).

#### *1.1 Biodiversity Action Planning (BAP: Victoria)*

The purpose of bioregional biodiversity action plans is to summarize the key biodiversity assets of the bioregion, and the actions and tools that are required to achieve statewide biodiversity goals. They are intended as preliminary plans to stimulate discussions and action planning within the regional community and to identify options for intervention that the local community can select from. A *Bioregional Strategic Overview* (e.g. Lowe et al. 2002) provides details of the methods used in developing a BAP and an overview of the features and assets of the bioregion. BAPs provide the bioregional context for the development of Landscape Plans and Local Area Plans. Landscape Plans provide specific information on assets and priorities for actions within parts of the bioregion and identify the best options for restoring native vegetation to recover biodiversity at a more detailed scale than is possible in the Native Vegetation Management Plans (developed under the Victorian Native Vegetation Framework). I wasn't able to get hold of any Landscape Plans.

In my reading of two Bioregional Strategic Overviews (Lowe et al. 2002, Anderson et al. 2003) and the BAP guidance for practitioners (Diez 2002), it appears that the BAP delivers a sound framework for identifying key values, assets and threats, compiling data, engaging the community in the planning process and educating people about general ecological and biodiversity management principles. The strategic overviews go to great lengths to describe the various legislative and policy mechanisms operating in each area. The BAP process does not provide a particularly compelling framework within which to prioritize areas of concern for conservation action or to rank a set of competing/candidate actions for investment in terms of their efficiency or contribution to overall biodiversity enhancement. There doesn't appear to be any specific guidance on how to assess the level of threat posed by particular threats or the urgency with which action is required. The Bioregional Network Analysis (BNA see section 3.4) method is used in BAP to identify general classes of actions appropriate for particular parcels of land depending on the threats faced by specific taxa in those places, though it falls well short of providing a framework by which to rank the catchment-level implications of particular actions/threats when compared with competing methods such as NSW's Biodiversity Forecasting tool (see 4.4).

### *1.2 Conservation Action Planning (CAP: International)*

The Nature Conservancy's Conservation Action Planning (CAP) process is designed to help conservation projects develop strategies, take action, and measure their success and then to adapt and learn over time. The CAP process covers the components of the Conservancy's conservation approach *after* global and ecoregional priorities have been set. The CAP is very much a project management tool. It provides almost no specific guidance on exactly how to identify and rank key values, priority locations, and candidate actions, other than to say that all these things are important and should be done. The CAP sets out 10 steps to successful project planning and management under four general headings of defining the project, developing strategies and measures, implementing strategies and measures, and adapting and improving. To its credit, the CAP is very strong in its requirement that project managers have a very clear idea of what project success means and what will be monitored to gauge that success. Surprisingly, there is very little guidance under the CAP system about what should be considered when deciding on, or ranking, competing potential projects. When referring to the identification of potential actions, the CAP states that; "Strategic actions are sets of interventions that you and your partners will undertake to achieve your stated objectives. *Your* challenge is to identify the high leverage actions that will enable you to get the most impact for the resources you have. There is no set formula for developing good actions other than using your situation analysis, asking probing questions to surface potential actions, evaluating the options, and then selecting for implementation those actions that are most promising and cost effective." There seems to be a gap in the CAP framework in terms of guidance on identifying priority areas and relevant actions. This could be addressed by integrating into CAP some of the assessment and evaluation approaches presented in sections 3 and 4.

## 2. Identifying goals, values, assets and threats

In this section I identify a number of key tools for use in defining and (sometimes) mapping the environmental assets and threats in a region. None of the tools reviewed fulfil all of the roles in identifying values, goals, assets and threats. Most of the tools focus on one aspect (e.g. assets). Indeed, tools for identification of assets (e.g. threatened species and vegetation communities, key habitats) far outnumber tools for assisting in goal setting or identifying threats. Explicit recognition of the values (*sensu* Wallace 2007) underpinning goals of biodiversity investment is rare and I could find no tools or frameworks to assist in this process (but see Wallace 2006, 2007). The tools to identify assets vary widely from vegetation condition mapping, to lists of threatened species, to wildlife habitat modelling and mapping methods. Many of the tools and resources mentioned here provide the input data to the prioritization tools and methods discussed in the next section. In many circumstances, environmental values will be identified by local community members and experts who have worked in the region.

### *2.1 Actions for Biodiversity Conservation (ABC) database (Victoria)*

ABC is a web-based information system to store, update and retrieve information about actions to recover threatened species and communities. The purpose of ABC is to identify priority locations for threatened species and communities and priority management actions at those locations, communicate actions and priorities to land managers, monitor progress towards implementation by recording and reporting on results, prepare and review Action Statements and Recovery Plans, and record and report on the state and trends for threatened species & communities.

ABC is an admirable and ambitious project in that it aims to store information about which actions were undertaken where in the landscape, allowing, in theory at least, for those data to be used to determine which actions are influencing trends in biodiversity. The actual mechanism by which temporal trends in species abundance and occupancy will be modelled, or the approach to evaluating the contribution of particular conservation actions to those trends is not yet finalized.

ABC is also described as a prioritization tool. The philosophy for prioritization embedded in the ABC is that priority should be afforded to locations in which a species is most likely to persist in the long-term (usually because the population is large, the habitat is secure, etc…). Priority is afforded to actions that, if implemented correctly would contribute most to reducing the risk of decline or extinction. ABC does not consider economic efficiency or the probability that the action will be successful (see 4.1), though there appears to be no reason why extra considerations couldn't be introduced by a given user. Apart from the general principles, there is little information about exactly how particular actions for conservation are compared and ranked within ABC. More detail on the actual ranking algorithm is being sought.

The DSE's BioSites database is another data base storing information on known sites of biological significance. It documents the biological assets present at those sites and what is known about potential threats and management requirements at each site. Typically significant assets are records of threatened species, vegetation communities, or suitable or potentially suitable habitat for threatened species. Mapping of BioSites is incomplete and the information is not publicly available.

### *2.2 Vegetation mapping*
Native vegetation in Victoria is classified in to Ecological Vegetation Classes (EVCs) which are the basic mapping units used for biodiversity planning and conservation assessment at landscape, regional and broader scales in Victoria. Similar types of vegetation class mapping are available in all other states (e.g. Keith 2005). EVCs are based on the types of plant communities and forest types (including species and structural information) and information that describes variation in the physical environment (topography, geology, landform, rainfall, salinity and climatic zones). Floristic communities within each EVC tend may respond in a consistent way to environmental factors such as disturbance (e.g. wildfire). As well as representing plant communities, EVCs may be useful as a guide to the distribution of many individual species and groups of species, including animals and lower plants such as mosses and liverworts. EVCs are often used as the basis for conservation significance mapping, though the quality of mapping varies geographically.

### *2.3 Condition mapping (Victoria, NSW)*
The concept of vegetation condition has recently gained currency in natural resource management policy in Australia (Keith and Gorrod 2006). Vegetation condition is now used for trend monitoring, measuring the impacts of development or management (ANZECC 2000), determining incentive payments (e.g., Eigenraam et al. 2006), vegetation clearing approvals, and determining appropriate offset actions (e.g. Victorian Government 2002). The concept of condition mapping has been extended to riparian and in-stream condition mapping, including the development of Victoria's Index of Stream Condition (ISC) metric.

Quantitative metrics of vegetation condition have been developed including Habitat Hectares (Parkes et al. 2003) in Victoria and BioMetric (Gibbons et al. 2005) in New South Wales. The objective of Habitat Hectares is to assess how 'natural' a site is by comparing it to 'benchmarks' representing the characteristics of mature stands of native vegetation of the same community type in a 'natural' or 'undisturbed' condition. Habitat Hectares assessments commonly include an assessment of the spatial context of the location using the *Landscape Context Analysis Tool* (Ferwerda 2002). This tool can be used in isolation to identify patches of vegetation that are more or less connected to other vegetation patches in the landscape, implying something about the viability or functionality of patches. A positive feature of Habitat Hectares and BioMetric is that they facilitate quantitative comparisons of protection actions across vegetation types, providing a single currency for all vegetation types. Both Biometric and Habitat Hectares aim to make assessment of vegetation condition more repeatable and transparent than other methods that rely on subjective judgement (McCarthy et al. 2004). Both approaches have made vegetation condition assessment substantially more tractable for a range of users than any previous rigorous approach.

Criticisms have been directed towards these schemes, including the subjectivity of some scoring components of the indices (Gorrod 2006), the inability to accommodate appropriate disturbance regimes, ambiguity surrounding operational guidelines for implementing the method and problems with the internal consistency of calculations of the vegetation condition score (McCarthy et al. 2004). Additive scoring systems can result in some unwanted outcomes (Gibbons & Freudenberger 2006). For example, an increased contribution of one attribute can compensate for the loss in another. McCarthy et al (2004) give an example in which the score lost from trees being cut down can be compensated by the increase in coarse woody debris. This reinforces the need for users to familiarise themselves with the component scores, rather than just overall site scores. Vegetation condition is difficult to measure and map over large areas and Habitat Hectares and BioMetric are a potentially useful way to approximate aspects of vegetation condition.

The measurement of vegetation condition is an important aspect of conservation assessment, when combined with threat level and habitat provision. However, care should be taken to avoid conservation assessments based predominantly on quality as this will tend to bias conservation away from the most threatened vegetation types that tend to be in worse condition than those that are less threatened.

### *2.4 Habitat modelling and mapping tools (International)*
Habitat mapping is the process of developing maps that identify suitable areas of habitat for particular flora or fauna species. A daunting array of habitat mapping methods exist, ranging from using experts to identify areas they believe to be good habitat, to expert derived rule-sets for habitat delineation, to highly sophisticated statistical modelling methods that utilize maps of environmental variables and biological survey data stored in GIS

An important feature of habitat modelling and mapping is that it is equally well suited to developing maps of biological threats, such as weed expansion prediction maps, feral predator maps. This is one of the highly under utilized features of habitat modelling that is only recently started to gain prominence.

While there are several specialized habitat modelling packages (e.g. MaxEnt; Philips *et al.* 2006) there are, unfortunately, no dedicated modelling packages that completely remove the technical trauma from habitat modelling. Nonetheless, habitat modelling is central to most modern, defensible biodiversity prioritization approaches. Some reviews provide tutorials aimed at lower-level users (e.g. Wintle et al. 2005), though generally speaking, NRM groups wishing to develop habitat maps for their region should pursue partnerships with agencies or universities equipped with appropriate skills and technologies. In some instances, habitat maps for a range of species may have been produced under particular planning initiatives (e.g. the Regional Forest Agreements) that may be of use to NRM groups in finer scale prioritization.

### *2.5 Biodiversity planning assessments (Queensland)*
Biodiversity Planning Assessment (BPA) is the implementation of the Biodiversity Assessment and Mapping Methodology that results in a map and database information product maintained by Queensland’s EPA (EPA 2002). The digital coverage results from a process of information collation, integration, analysis, interpretation, spatial data development and mapping. Biodiversity Planning Assessments identify three levels of Biodiversity Significance - State, Regional and Local - based on a number of data queries that simultaneously integrate an array of information for a bioregion. They may also indicate areas that have not been assigned a Biodiversity Significance because they have not met the criteria for State, Regional or Local Significance based on current information. The method

employs a number of routinely measured criteria; remnant units, essential habitat (for EVR species), ecosystem value, tract size, size relative to ecosystem, condition, ecosystem diversity, and context and connection. Each landscape unit receives a ranking for each criteria; low, medium, high, very high. Criteria are then combined with ‘and/or’ statements to determine a ‘first-cut’ priority ranking of sites. An expert panel then refines those prioritizations. The actual use of the maps (reason for their construction) is unclear. The method is well documented (EPA 2002). The decision to arbitrarily categorise criteria rankings is somewhat bemusing, given that each are based on well chosen and well measured quantities (e.g. the % of total area of an ecosystem comprised by the land unit). The resulting rankings are therefore somewhat opaque (ie. exactly why a place receives a particular rank is not easily determined at first viewing. Nonetheless, the data collation and storage effort is appealing and will be a valuable resource for NRM biodiversity planning in Queensland.

## 3. Identify spatial priority zones and candidate actions

There are a growing family of spatial prioritization tools in the literature. The bulk of tools have been developed for static allocation of lands between reserve and non-reserve land tenures (e.g. Marxan, C-Plan, Zonation), while recent extensions enable allocation of lands between multiple tenures or land-uses (e.g. Marzone, Oprah). Here I review a few of the most widely used of the spatial prioritization tools.

Much of the spatial prioritization literature describe spatial prioritization tools as though they are providing ‘the decision’ about where and what to do in the landscape. Here, I review these tools in terms of how well they can identify or highlight spatial zones or groups of sites as being of particular importance to biodiversity. This is based on the assumption that a separate process of identifying and ranking potential actions for priority zones will ensue. There is very little guidance on appropriate processes for identifying candidate actions once priority areas are identified. In order to make the descriptions of prioritization tools compact some of the detail on some methods has been removed to Appendix 1.

### *3.1 Marxan/Marxan with Zones (International)*

The MARXAN software (Ball and Possingham 2000; Possingham *et al* 2000) was developed in the late 1990s to provide decision support for reserve system design in marine and terrestrial systems. MARXAN attempts to meet a suite of user-defined biodiversity targets for reserve selection problems, finding solutions to the section of spatially cohesive sites at minimum cost. Costs may include the cost of purchase, cost of management; any cost that is estimable. Unlike Zonation (see 3.3) which provides a nested set of solutions, MARXAN optimizes for a pre-specified target (e.g. 30% of landscape conserved).

MARXAN is the most widely used conservation planning tool, so it is a well known and understood method. Advantages of MARXAN include the ability to incorporate costs, and to generate numerous landscape planning options. Output is in the form of maps that can form a compelling part of stakeholder negotiation. Both programs are simple to run, and robust enough to be applied to a multitude of different systems.

A frequently cited weakness is that MARXAN takes a very simplistic view of the implementation process, by assuming that every important factor can be included as either a feature or a cost. To that extent, MARXAN is more of a prioritization-support tool rather than a prioritization method. It cannot tell conservation managers exactly where reserves or conservation zones should be placed, but can offer landscape plans which would achieve biodiversity goals at a minimum cost. These plans can then be taken to stakeholder groups for negotiation.

The primary limitation of MARXAN for non-reserve design planning was that it could not deal with multiple land-zoning or land management alternatives. Hence the development of

‘MARXAN with Zones’ (or MARZONE). MARZONE is capable of allocating land to a series of different classes (either tenures or just management priorities).

The MARZONE and MARXAN methods also assume that conservation can be implemented instantaneously, and do not explicitly consider possible dynamic threatening processes. MARXAN and MARZONE indicate which planning units should be included in conservation plans, but cannot indicate which ones should be sought after first, if all are not immediately available. This may not be a problem if plans can be immediately implemented in their entirety. This may be problematic when managers do not have the funding to instantly purchase or invest in management of all the necessary land.

MARXAN is available free of charge from The Spatial Ecology website http://www.ecology.uq.edu.au/marxan.htm where links to the MARXAN manual can also be found. MARZONE is not yet formally released. It is hoped that the program and the user manual will be available by the end of 2008. See Appendix 1 for more detail.

### *3.2 C-Plan (International)*

C-Plan (Conservation Planning System) (Pressey and Logan 1995) is another example of a software tool designed to support place prioritization and conservation planning decisions. C-Plan was primarily developed for use in the New South Wales Comprehensive Regional Assessment as part of the Regional Forest Agreement process (Commonwealth of Australia 1999). C-Plan uses a similar approach to MARXAN with respect to achieving a set of conservation goals for as many features as possible in the minimum area, however, unlike MARXAN, it does not offer a globally optimal (or near-optimal) solution to reserve selection with multiple constraints and considerations (such as boundary-length penalty and spatial spread).

One of the key pieces of information that C-Plan calculates and displays is the irreplaceability of each site in the planning region (Ferrier et al., 2000). The irreplaceability of a site can be used as a guide to the importance of that site for achieving a regional conservation goal. Highly irreplaceable sites are crucial for achieving the goal, and failure to select them usually means that the goal for one or more features cannot be achieved. Sites with a low irreplaceability are less crucial because there are other sites that can contribute equally or greater to the achievement of the conservation goal. In other words, sites with a low irreplaceability indicate that there are many options about which one to choose, although some of them must be selected if the regional goal is to be achieved.

C-Plan links directly with MARXAN to enable global searches for optimal reserve configurations. C-Plan can also connect to another module called SPATTOOL which guides the achievement of spatial configuration objectives, allowing explicit consideration of *patch size and connectivity, and geographical and environmental spread*. C-Plan is Windows-based and links to a GIS. The C-Plan software and user manual can be downloaded from http://www.uq.edu.au/~uqmwatts/cplan.html. See Appendix 1 for more details.

### *3.3 Zonation (International)*

Zonation is a spatial conservation planning framework that utilizes species distribution model predictions to identify areas important for retaining habitat quality and connectivity for multiple species (Moilanen et al. 2005). Zonation produces a hierarchical prioritization of the landscape based on the conservation value (habitat value) of sites (cells), iteratively removing the least valuable cell (accounting for complementary) from the landscape until no cells remain. The program is relatively simple; starting from the full landscape, the program determines the site (or cell) in the landscape with the lowest marginal value and removes it. This process is continued until all cells are removed. The order of removal identifies the relative priority value of each portion of the landscape. Zonation operates on a set of raster maps of species habitat, one per species. Each cell value represents either an observation of population size at that location, or a probability of occurrence or abundance predicted using a statistical habitat model. Point occurrence data can also be entered. Point distribution data, planning units, species interactions, directed freshwater connectivity and several other types of data can be included to allow incorporation of costs and other constraints on the solution.

Importantly, Zonation cannot handle multiple alternative land-use options (as in MARZONE) or multiyear incremental (dynamic) design of reserves (but no program does that well yet). However, Thompson et al. (in review) have developed a strategy for using Zonation to identify priority areas for vegetation restoration based on the predicted *future* value of areas for fauna species, assuming that they were successfully restored to native vegetation. Their method purportedly identifies a set of locations in which restoration action will produce a 'balanced' set of species representation over time.

Zonation is only available for Windows and it does not work with vector maps as does MARXAN and CPlan. It only allows a limited set of interactive planning analyses. Zonation is free and can be downloaded software from the Zonation web pages: http://www.helsinki.fi/bioscience/consplan. The software package includes user manual and tutorial. See Appendix 1 for more details.

### *3.4 Bioregional Network Analysis (BNA: Victoria)*

BNA is a system for bioregional ranking of threatened flora and fauna used under the Victorian Bioregional Biodiversity Action Planning (BAP) framework. The bioregional status and ranking (required response level) for each taxon is determined through five steps: (i) estimating the percentage of the State population which occurs within the bioregion (based on records on NRE flora and fauna databases), (ii) estimating the percentage occurrence in each land tenure in the bioregion (based on the databases plus local knowledge and expertise), (iii) allocating a ranking (1, 2, 3 or 4) for each parcel, based on the importance of its contribution to the overall occurrence of the parcel within the state, (iv) estimating a risk ranking (A, B or C) for each parcel, based on the statewide conservation status of the taxon and its estimated population trend in this parcel, (v) allocating an *expected response* level for action (i.e. 1 for highest priority to 4 for lowest) for each parcel based on the combination of the Occurrence and Risk ranking. Guidance as to the type of response that is required at each of the expected response levels is provided in a table (see Fig. 1).

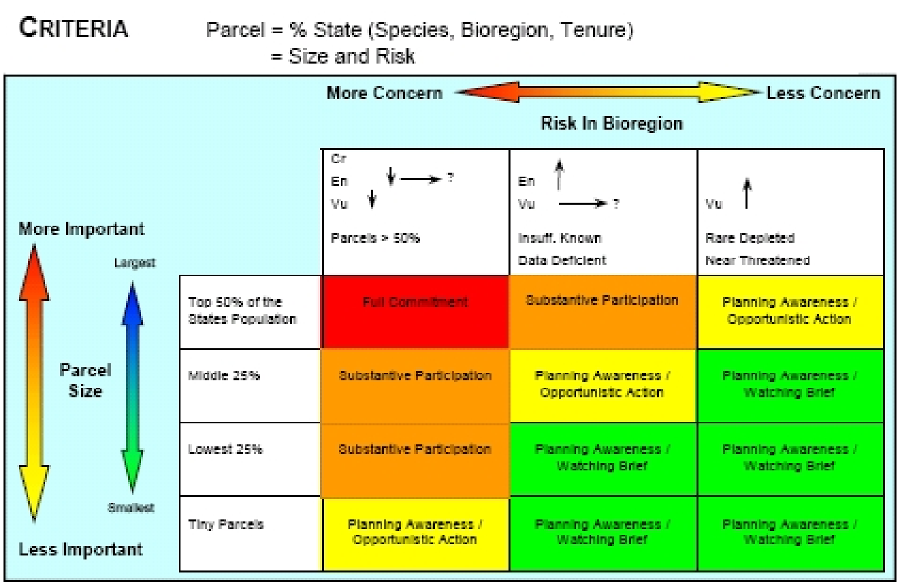


**Figure 1.** A risk matrix scoring system for Bioregional Network Analysis. The risk table is used to identify general classes of actions appropriate on parcels of land within a bioregion according to five general principles outlined above.

BNA can be viewed as a rule-of-thumb tool that doesn't link directly to any particular model for how the species responds to disturbance or how changes to the state of the particular parcel of land influences the probability of persistence for any particular taxa.

### *3.5 Optimal restoration of altered habitat (OPRAH: South Australia - at present)*

Westphal et al. (2003) adapted the core simulated annealing algorithm in MARXAN and developed an approach for the optimal design of terrestrial landscape rehabilitation strategies based on target species responses to landscape metrics. OPRAH expands on the work of Westphal et al. (2003) by allowing habitat quality and species assemblage rules to also be considered in the design. In contrast to MARXAN, which divides the environment into GIS polygons, OPRAH uses GIS raster grids. Budgetary constraints, costs and land use restrictions are important considerations in any rehabilitation planning. OPRAH incorporates information about extant native vegetation, land use and economic data. OPRAH is capable of solving problems of maximum restoration budget for multiple species, and single-species reconstruction objectives for off-reserve conservation planning. A beta version of the software is available on request, but OPRAH will not be freely available with manual and tutorials until early 2009.

## 4. Prioritizing/ranking a set of actions

Having identified a set of spatial priority areas using any of the tools described in 3.3, a series of tools exist to assist in the development of project priorities that address specific assets and threats in the high priority locations. For example, having identified some key habitats in a region, NRM groups may wish to consider which restoration or protection projects are most appropriate given the specific assets and threats contained in, or potential acting on those habitats. This would potentially give rise to a list of projects (such as fencing, weed eradication, incentives for grazing reduction, ecological burning, etc..) that could potentially be undertaken. There are no tools *per se* to assist in the identification of specific candidate management actions for a particular location, asset, or threat. Tools such as BNA and ABC provide some general guidance as to the types of actions that are appropriate in very general classes of area or to deal with general threat categories, though the actual identification of specific projects remains the domain of local experts and community members. If the list of projects identified is more than can be supported in a given years budget, it may be desirable to rank competing projects according to their urgency, feasibility and cost. This section provides a brief summary of some of the tools available to assist groups undertake such a ranking exercise.

### *4.1 Project prioritization protocol (International)*

The 'project prioritization protocol' (PPP) is a simple rule-of-thumb first described by Possingham et al. (2002) for prioritizing national-level investment in biodiversity projects. The protocol aims to formalize three key elements of prioritization; probability of project success (p), expected benefit conditional on success (b), and expected cost (c) in order to identify a cost-efficiency measure (E) for biodiversity projects:

$$E = (p*b)/c \qquad \text{(Eq. 1)}$$

Joseph et al. (2008, in press 2008) further develop the idea and apply it to prioritize investment in species recovery projects for listed threatened species in New Zealand. Expected benefit (estimated by experts) was defined in terms of the change in predicted risk of extinction arising from the implementation of a proposed project compared with the risk faced were the project not implemented. A more rigorous approach to estimating benefit could be employed using population viability analysis models (see 4.4).

PPP provides a coherent approach to prioritizing projects. Strangely, it is the only prioritization method reviewed here that avoids the use of arbitrarily scaled indices (of benefit, threat, or feasibility), that explicitly incorporates $ cost of implementation, **and** that explicitly considers the probability of success. PPP is very simple to understand and explain. On the downside, PPP makes no attempt to incorporate complementarity between projects (cf with *Biodiversity Forecasting*) and difficulties associated with estimating highly uncertain parameters such as probability of success or expected benefit apply to this method, just as they apply to other methods described here.

An additive index developed by Marsh et al. (2006) provides an alternative approach to prioritizing threatened species for management projects and is currently being trialled by the EPA in Queensland. The index provides a multi-attribute ranking scheme whereby numerous criteria such representing aspects of uniqueness, social importance, biological importance, and potential for recovery are given scores between 1 and 4. Scores are then combined to provide a ranking for each species that can be used to determing restoration expenditure. The scoring system requires experts to assign scores to the individual criteria. Because the scores are not ‘measurable’, it is difficult to see how such a system could be embedded in an adaptive management framework. Similarly, because scoring is additive, it is possible for a project that scores very highly on social importance but very poorly on feasibility (ie. is almost completely infeasible) to be ranked highly. This sort of problem arises routinely in additive scoring systems. However, the simplicity of the scoring system has appeal and presuming the number of candidate projects is small enough, logical filtering could deal with some of the numerical problems.

### *4.2 Biodiversity benefits index (Oliver and Parkes 2003: NSW, Victoria)*

Oliver and Parkes (2003) developed a *Biodiversity Benefits* toolkit that built on the Habitat Hectares approach to assessing vegetation condition. The toolkit is targeted towards evaluating biodiversity benefits (or dis-benefits) of a given action or land-use change. The benefit index aims to capture the requirements of a broad range of flora and fauna rather than the specific requirements of individual (rare or threatened) species. It scores *benefit* according current vegetation condition, the conservation significance of the site, and the landscape context of the site. The benefit of a proposed action is then the difference between the existing (or no-action) benefit and the benefit score predicted after the action is implemented. In this way, it is a non species-specific version of PPP (above) without the explicit incorporation of feasibility (probability of project success). It would be a short step to incorporate feasibility and cost in the BBI methodology, and indeed, the inclusion of cost is discussed in the original manuscript (Oliver and Parkes 2003).

The contributing components used in the BBI (with the exception of area) are arbitrarily scaled indices (ranging between 0-100 or 0-10) rather than measurable values such dollar cost, probability of success, and probability of extinction of population decline. This is largely unavoidable given the difficulties associated with combining the many aspects of vegetation condition or functionality into one 'measure' that can be used to compare the outcomes of competing management options. Further details in Appendix 1.

### *4.3 Biorisk (Western Australia)*

One component of Biorisk (K Wallace *pers. comm.*), a new project within the Future Farm Industries Cooperative Research Centre, is a decision process for biodiversity assets threatened by altered hydrology. This decision process aims to integrate management planning and decisions from high order values, goal setting and asset specification on the one hand; through to feasibility analysis. While methods have not been fully developed and trialed, evaluation of risk within the feasibility component will include a range of techniques including subjective risk assessment protocols and fault-tree analysis methods developed by Burgman (2005) and Walshe et al. (2005, 2007). Overall, the decision process builds on the management framework outlined in Wallace et al. (2003). Subjective risk assessment approaches utilize risk matrices and fault trees that provide a visual representation of the conceptual modelling underpinning the risk assessment (Appendix 1).

The development of the risk matrix requires the prior identification of relevant values, goals and assets. The risk matrix, which is initially based on expert assessment, describes the probability that specific threatening processes will lead to goal failure. For those threats that pose a high probability of goal failure cause-effect relationships, and ultimately feasibility and certainty of management success, are initially explored using cause-effect tools (such Bayesian Belief Networks or fault trees) followed by more complex numerical techniques once enough data is obtained.

Fault trees are a useful subjective risk-assessment tool for assisting experts and stakeholders in documenting ideas of cause and effect and provide explicit descriptions of uncertainty (Appendix 1). Fault trees also assist users to coherently (if not precisely) estimate the likelihood of a given outcome using ordinary probabilistic calculus.

Subjective risk assessment methods are a potentially powerful tool for estimating and ranking the benefits of biodiversity investments where little or no data and/or appropriate expertise exist for fitting more detailed or predictive models. Even where such data and expertise do exist, risk matrices and fault trees may still serve as very useful communication devices. Of all the tools described in this review, only risk assessment methods (and methods described in the next section) provide the opportunity for formal exploration of uncertainty.

### *4.4 Population models for scenario analysis (International)*

Natural resource managers may seek information about the likelihood that native species will persist for a given time into the future within a network of habitat patches given a certain management regime (Flather et al., 2002). This information can be provided by population models that permit a detailed mechanistic representation of population dynamics and the processes that make a population vulnerable to decline or extinction (Boyce, 1992). While there is considerable uncertainty associated with using population models to predict actual risks of extinction (Fieberg and Ellner, 2000), population models can be useful for predicting changes in risks of extinction and for ranking different management strategies (McCarthy et al., 2003). Important strengths of population models include transparency (embodied in explicit assumptions; Burgman and Possingham, 2000), freedom from linguistic ambiguity and the ability to incorporate stochasticity and other forms of uncertainty (Wintle et al. 2005). Managers can make decisions based on probabilistic statements, for example "the risk of extinction in 50 years under management action A is 0.6 while under action B it is reduced to 0.3". Results are repeatable and internally consistent (Burgman et al., 1993). Furthermore, the actual process of constructing a model can be useful as it provides a framework for compiling relevant information, articulating expectations and management options and examining uncertainties (Burgman and Possingham, 2000).

The modelling package RAMAS Landscape (Akçakaya et al., 2003) integrates spatial models of landscape change (also known as landscape dynamic models or vegetation succession models) with species population models, enabling incorporation of temporal processes such as climate change and vegetation restoration. All population models, including RAMAS Landscape models have relatively high expertise and data handling requirements and would generally only be useful for analysing the influence of management actions for a few iconic species in a region. The construction of DLMP models would generally require a collaboration between NRM groups and an agency or university.

### *4.5 SCARPA/Biodiversity Forecasting (NSW)*

*SCARPA* (Site and Catchment Resource Planning and Assessment Decision Support system) is part of an integrated software application to support catchment planning and investment decisions in NSW. *SCARPA* uses spatial and other data to drive a suite of biophysical models that predict the outcomes of land use or management change on biodiversity, land and soil capability, salinity, and carbon sequestration.

*SCARPA* has a *Biodiversity Forecasting* module which is a GIS-based approach to regional conservation assessment. *Biodiversity Forecasting* has been designed to consider the contributions, both positive and negative, that multiple types of land-use and management (not just designated "conservation areas") make to maintenance of biodiversity across a landscape.

*Biodiversity Forecasting* does not require a predefined set of "conservation targets" specifying the area, or amount, of each biodiversity entity (e.g. species, community) to be conserved. The approach is used to evaluate the landscape level impacts of different management scenarios. Spatial data, expert opinion and models for the extent, condition and configuration of vegetation types (or habitat for individual species), pressures/threats, and existing or proposed land use/management, are integrated into *indices* of current or predicted (future) regional biodiversity status. Regional conservation status can be assessed in relation to either biodiversity as a whole, using vegetation communities as a general surrogate for terrestrial biodiversity, and/or individual species of particular conservation concern including threatened species. At the individual species level, the approach is better suited to animals than plants (Ferrier 2005).

Without having run the *Biodiversity Forecasting* tool, my impression is that it is a relatively sophisticated tool for predicting the catchment/landscape level impacts or benefits of any action that influences native vegetation and habitat. It appears to demand fairly sophisticated user interactions to specify response functions of priority species to particular actions. Unlike MARXAN or MARZONE it is not a global optimizer (ie. it won't identify optimal actions for particular places). However, it does appear to be a particularly powerful tool for investigating the catchment-level implications of particular management actions or biodiversity investment decisions. There is substantial documentation in technical reports produced by DEC for particular planning exercises (DEC NSW 2006a,b) and a user manual is under preparation (Tom Barrett pers. com.).

### *4.6 Strategic landscape investment model (SLIM; Hajkowicz et al. 2005: NSW, QLD)*

The strategic landscape investment model is a tool for mapping optimal environmental expenditure. This tool was developed in NSW to assist in prioritizing investment in establishing perennial pasture on grazing lands. The tool produces maps of marginal environmental benefit per dollar spent. These maps are used to define an optimal treatment design within a budget. The landscape attributes currently considered include salinity, water yield, nitrogen run-off, phosphorus run-off, stream sediment concentrations, soil erosion and carbon sequestration. The tool hasn't yet been used to incorporate biodiversity, though there appears no reason it couldn't be included. The optimization is not spatially explicit in the sense of considering non-linear relationships between chosen options. It appears to only handle land-use change, rather than continuous investment options or other actions. Like PPP, SLIM is appealing because optimizes investment using measures of real quanta (e.g. change in phosphate run-off in mg/l, or change in water yield in ML/year) rather than arbitrarily scaled indices. SLIM warrants further investigation.

Victoria's EcoTender scheme uses a Catchment Modelling Framework (CMF) not dissimilar to SLIM that integrates multiple biophysical attributes (see Appendix 2 and refs therein).

### *4.7 Biodiversity Prediction using Ecological Processes (BioPrEP )*

Biodiversity Prediction using Ecological Processes (BioPrEP) (Mackey *et al*. 2008) is underpinned by a hierarchical framework of goals, criteria and indicators: goals are the conservation outcomes the user seeks to achieve, criteria are standards by which each goal is judged, and indicators are specified to measure the condition of each criterion. BioPrEP integrates spatially-explicit datasets that represent the conservation goals in a GIS environment with a decision-support tool to prioritize parcels of land for conservation investment. The decision-support tool (Multi-Criteria Analysis Shell - Spatial [MCAS-S]; Bureau of Rural Sciences 2007) has been designed so that technically unskilled operators can intuitively and easily interrogate land assessment data to explore natural resource management investment options. MCAS-S assumes the user has already generated the necessary indicator values using a GIS.

The primary analysis is a spatial comparison of the indicators used to represent the conservation goals. This creates an “area of interest” layer defined by the spatial coincidence of the desired levels of each of the indicators. The results are then imported into a GIS to map the “area of interest” in relation to relevant land parcels (e.g. pastoral properties or subcatchments). The land parcels are then ranked according to the cumulative extent of “area of interest”. This structured ranking of land parcels provides the basis for defining the range of investment options. A reporting output consists of a ranking of land parcels. Costs of land acquisition and management may be included as additional criteria. The primary limitations of BioPrEP are the availability and generation of spatial data for the indicators chosen to represent the biodiversity goals. BioPrEP doesn't appear to account in any way for complementarity between investment options, so the decision to choose a particular investment option would not necessarily influence the ranking of the remaining options. BioPrEP is currently in development.

BioPrEP has been trialed by Bush Heritage Australia the goals of: (i) capturing source (naturally productive) areas; (ii) protecting areas with the highest remaining functional integrity; (iii) improving the level of protection of the least protected ecological types; (iv) protecting functionally viable populations of significant species or assemblages and their (biophysical) habitat; (v) contributing to mitigation of current and future threats to Australian biodiversity; (vi) spreading investments across bioregional gradients; and (vii) optimizing spatial configuration of protected habitat.

## 5. Discussion and conclusion

There is a vast array of tools available to assist in the identification and ranking of biodiversity investment option, each tool has a slightly different purpose, differing data and expertise requirements and is more or less amenable to use with experts (Table 1). Here, I have classified tools according to which part of the biodiversity prioritization process they are best suited to (Fig. 2). The classification isn’t perfect, in the sense that some of the tools could conceivably be used in more than one of the three general steps in prioritization. Most of the methods for identifying assets, threats and goals are, and will continue to be useful in various NRM prioritization activities. However, there does appear to be a lack of well documented tools and data for identifying and mapping biological threats.

Very little attention has been paid in the biodiversity conservation literature to the socio-economic issues or processes that influence the adoption and implementation of conservation plans (but see Knight et al. 2006). There is precious little guidance for practitioners on how to choose appropriate policies or strategies for ensuring the success of biodiversity conservation or restoration plans. This is an area of conservation planning that is deficient and in need to further research. Similarly, there is little guidance on how to reach consensus about appropriate management goals. There is a need for further research and investment in social scientific methods for determining and setting goals for biodiversity conservation and the catchment level. This is particularly important because decisions at the outset of a planning process about goals and measures of success (e.g. improving persistence probabilities of threatened species or improving vegetation condition) will determine the sorts of actions that are proposed and, in turn, lead to very different outcomes.

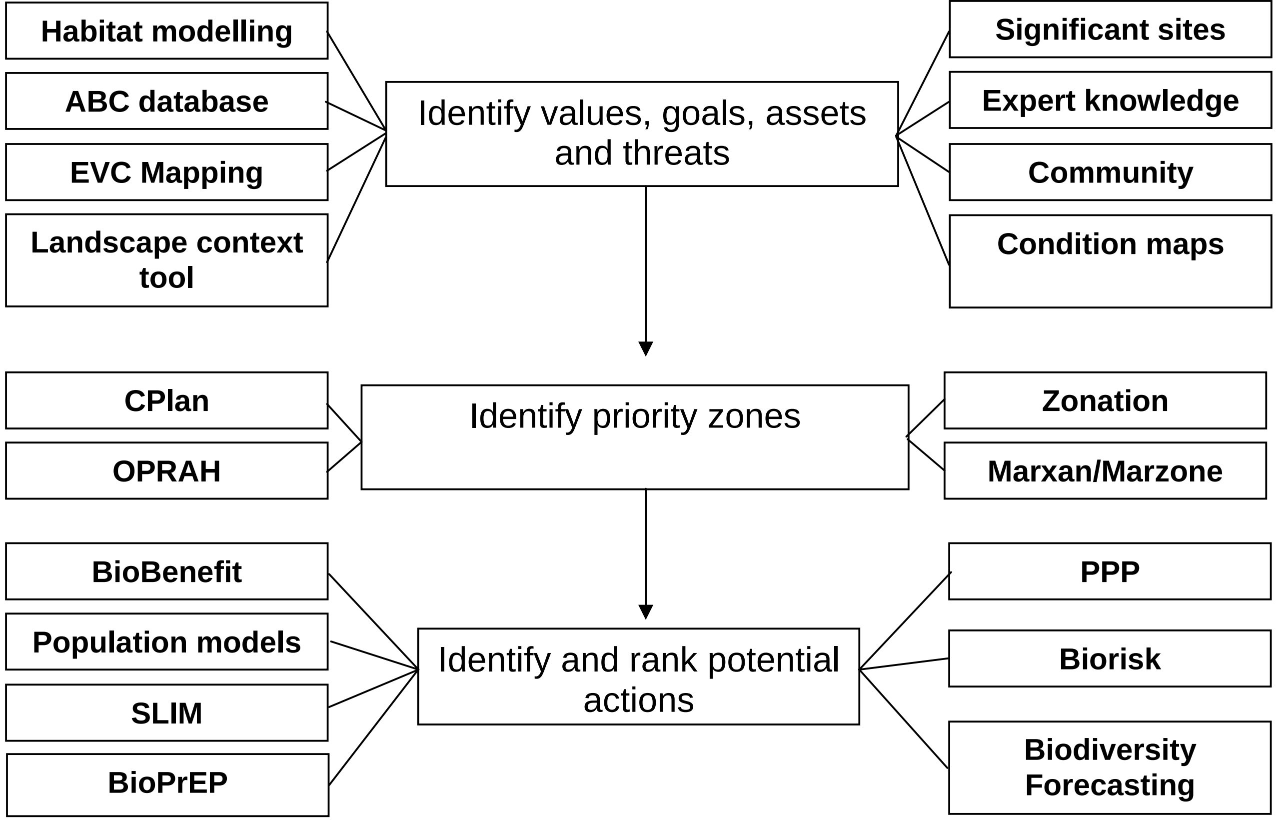


**Figure 2.** An inventory of tools for prioritizing investment in biodiversity conservation and restoration projects. Here I have identified three general steps of the prioritization process: (i) identification of values, goals, assets, and threats, (ii) identification of priority zones in which high concentration of high value assets exist, and (iii) identification and ranking of options for acting on specific projects or dealing with specific threats in the priority zones.

A key disadvantage of most spatial prioritization tools is that they fail to deal explicitly with uncertainty in a systematic way. With the proliferation of methods and tools available, prospective users are faced with questions such as "why trust complex tools above intuition or basic rules of thumb?" Given uncertainties in underlying data, can practitioners have confidence in the plan and defend their plan in a politically charged decision making environment? Answering these questions entails the difficult task of predicting how a method will perform to a particular problem, or on a broad range of problems. Little indication is given about the consequences of violations of assumptions about species abundance and changes in land availability, cost and condition. It would seem that we are a long way from having tools that deal with uncertainty routinely (e.g. within the MARXAN algorithm). For the time being at least, risk assessment approaches (Burgman 2005) provide the most coherent strategies.

Of the spatial prioritization tools reviewed any of those presented would provide a sound basis for identifying general areas of high biodiversity priority within which particular assets and actions could be identified. MARXAN provides a well understood set of tools for identifying zones of importance and their use of polygon data is probably advantageous (cf Zonation and its use of raster data). However, Zonation has the appealing feature of not requiring the specific identification of conservation targets. A combination of PPP and Biodiversity Forecasting would provide a highly robust and logical approach to ranking projects and assessing the catchment-wide implication of choosing or ignoring particular options. While PPP is a very simple concept and easily implemented, it provides no indication of the complementary or context dependent value of particular projects. That can be explored very rigorously within the Biodiversity Forecasting framework. However, this is conditional on NRM groups being able to invest in the technical expertise required to implement the more complex methods. The subjective risk assessment approaches described by Walshe et al. (2007) and possibly emerging in Biorisk may be useful alternatives to the technically demanding forecasting tools or population models.

Market-based instruments (MBIs) are an important emerging force in biodiversity conservation. I haven't provided a detailed review of MBIs here because they represent more a general approach for generating conservation opportunities, rather than being a conservation planning tool *per se*. I have provided a brief review of a couple of key MBIs (Bush Tender and EcoTender) in Appendix 2. This review is not exhaustive. There are some biodiversity tools not reviewed in detail here and some previous research activities that utilize a variety of tools that may be of value in particular applications (e.g. MacKewan et al. 2004). Table 3 below to enable follow up research.

| Tool | Purpose | Contact |
|---|---|---|
| Land Use Impact Model | Spatially evaluate environmental threats using a risk assessment framework | (MacKewan et al. 2004) |
| MCA Shell | Integrates and explores spatial information | Rob Lesslie, Bureau of Rural Sciences |
| NRM Toolbar | Shares knowledge between regional NRM bodies and information providers | Mathew Silver, Land & Water Australia |
| Scenario Analysis | Quantifies the benefits of plans using performance indicators | David Pullar, UQ |
| Spatialise | Shows the best places to carry out management actions according to users' criteria | Rowan Eisner, NRW |

**Table 3.** Tools identified as potentially relevant to NRM biodiversity prioritization.


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## Appendix 1. More detailed notes on prioritization tools.

### 1.1 Marxan/Marzone

MARXAN can select between as many as 20,000 planning units using data on species, habitats and/or other relevant biodiversity features and surrogates. Key to the efficiency of Marxan is the focus on minimising costs while meeting user-defined biodiversity targets. MARXAN uses the mathematical method of simulated annealing to search through millions of candidate reserve systems, looking for the best reserve configuration. Its utility lies in the ability to consider a range of factors, in conjunction, when selecting a set of sites. These factors can include design features, such as minimising the boundary length (i.e. maximising compactness) of new conservation areas, representing features in clumps, and also in a minimum number of clumps separated by a specified distance (to minimise risk from catastrophic events).

A longstanding limitation of MARXAN was that it created binary conservation landscapes, with areas being either protected or unprotected. However, when these conservation plans are implemented, they generally plan for a set of conservation zones, only some of which are strict protected areas. For example, the final rezoning of the Great Barrier Reef created 7 different conservation zones, some of which could be used for commercial or recreational extractive activities (e.g., the Habitat Protection Zone prohibited commercial trawling, but did allow commercial line fishing, and harvest fishing for aquarium fish to permit holders). The solution to this problem was MARZONE, an extension of the original MARXAN program to allow for multiple conservation zones. These new capabilities mean that natural resource managers will be able to use MARZONE to identify configurations of sites that meet a range of management objectives whist minimising cost.

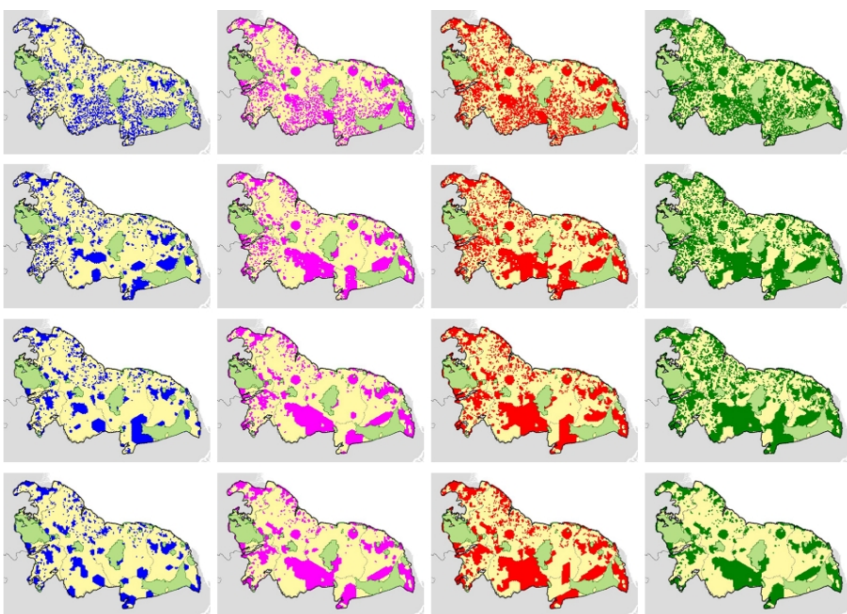

**Figure 1.1.** Example output from MARXAN demonstrating multiple options to inform WWF's plans for the moist rainforests of Guiana.

MARXAN is the most widely used conservation planning tool worldwide, with over 1100 users within 600 organisations, from 60 countries. Examples of the use of MARXAN include supporting the zoning of the Great Barrier Marine Park, design of marine reserves in the Channel Islands National Marine Sanctuary, and producing a set of spatial priorities in the Romanian Carpathians to protect large carnivores. Perhaps the best known application was to the 2002 rezoning of the Great Barrier Reef Marine Park in Australia, the world's largest and most comprehensively managed marine park. Other significant applications include

TNC's conservation plan for the Channel Islands of California and WWF's plans for the moist rainforests of Guiana (see Figure 1.1).

The optimisation algorithm that attempts to find good systems of sites is 'simulated annealing' (Kirkpatrick et al. 1983, Otten et al. 1989). One of the most useful outputs from MARXAN is the 'summed irreplaceability' output (Leslie et al. 2003, McDonnell et al. 2002). This output shows how often each planning unit is in one of the chosen reserve design systems. Planning units that are chosen more than 50% of the time can be thought of as being essential for efficiently meeting biodiversity goals. Sites that are rarely selected can be ignored. This concept is inspired by, but different from, Bob Pressey's notion of irreplaceability (Pressey et al. 1994).

MARXAN and MARZONE are based on static conceptualisations of the conservation landscape, and of the implementation process. For example, once protected, land retains its valuable conservation features perpetually. It is also assumed that the plans can be implemented instantaneously, and that all land is available for conservation unless specified as unavailable (the costs of the land can be different).

Both programs consider land to have "features", which will be maintained if appropriately protected, and lost of not. Features that have been protected contribute to regional conservation "targets" (e.g., 10% of all Victorian grasslands) in a linear additive fashion.

Both programs require considerable amounts of data (unavoidable, when planning a conservation system for an entire landscape). The landscape is discretised into a set of "planning units". The data required by MARXAN falls into two main categories – features in each planning unit, and overall feature targets. The spatial location of every feature of importance has to be provided to the program, including the amount of that feature, if this is relevant (e.g., the hectares of grassland in a particular planning unit). The amount of each feature that needs to be protected, the "target" must also be defined.

In MARZONE, one needs to define exactly how each type of conservation zone will affect each feature of interest. For example, if we had a marine conservation zone that allowed no extractive activities, except for line fishing, we would have to specify the impact of this activity on each of the features in that planning unit. So, we would have to say that line fishing does not affect coral species, or seagrass, but does negatively affect commercial fish species.

### 1.2. C-Plan

Pressey et al. (1994) proposed that the irreplaceability of a site be measured as the proportion of all representative combinations of sites in which that site occurs. Consider, for example, a situation in which a system of protected areas must be formed by selecting a combination of n sites from a total of t sites. The number, C, of possible combinations of size n that can be constructed from these sites is:

$$C = t!/n!(t-n)!$$

which is the binomial coefficient. The measure of irreplaceabilty used in C-Plan and MARXAN differ slightly from the original proposition, though the essential principle remains.

All calculations in C-Plan are based on a matrix of sites by features and are driven by feature targets (the area or number of localities of each species, forest ecosystem or other feature) identified as requiring some form of conservation management. The targets are updated each time one or more sites are selected. Updating targets changes the area or number of sites still needed for conservation according to their extent in the recently selected sites. In ArcView, sites are coloured according to whether they have been selected in C-Plan, or according to their current irreplaceability value. At any stage of a conservation planning exercise, you can produce detailed reports on sites and/or features. These reports can be read directly into a spreadsheet program such as Microsoft Excel.

### 1.3. Zonation

The order of removal of cells in the landscape is determined by the marginal value of a cell ($\delta_i$):

$$\delta_i = \max_j \frac{Q_{ij}(S)w_j}{c_i}, \qquad \text{(Eq. 1)}$$

where $Q_{ij}(S)$ is the proportion of the remaining distribution of species $j$ present in cell $i$, calculated for the set of remaining cells $S$, $w_j$ is an arbitrary weighting assigned to species $j$ (e.g. based on threat category), and $c_i$ is the cost of cell $i$ (either purchase cost, or cost to maintain).

Using this iterative removal technique, landscapes can be zoned according to their value for conservation. The program produces, among other things, basic raster files from each run, which can be imported to GIS software to create maps or to conduct further analyses. The data requirements for the program are realistic and it can be run with large datasets containing up to 4000 species on a 16 million element landscapes on an ordinary desktop computer. Zonation includes species-specific connectivity responses, and weighting of species. The analysis is deterministic and the results of a zonation run can be summarized as a single map with zones (see Fig. 1.2).

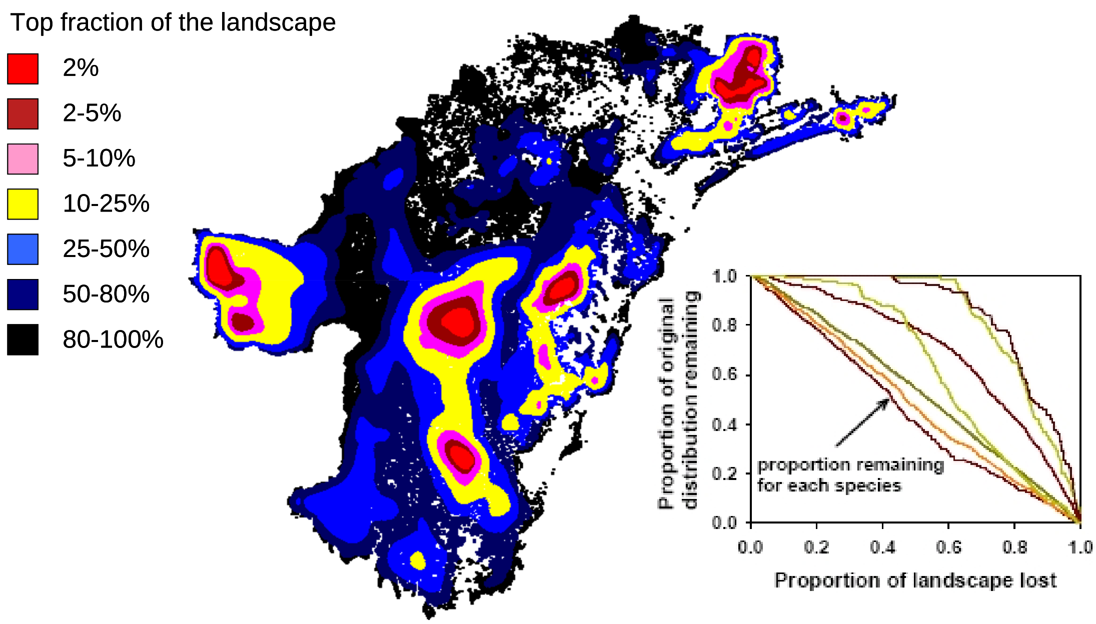


**Fig. 1.2.** Zonation produces hierarchical solution of conservation priority based on the benefit function algorithm (Eq. 1). The hierarchical solution is presented here in map form. The red areas represent the highest priority areas (top 2%). The black areas are the lowest priority areas (bottom 10%). Zonation also produces a graphical representation of the landscape trade-off for each species. Each species is represented as a line on the graph. This graph shows the proportion of the total habitat for each species remaining in the landscape as the solution proceeds from 100% conserved down to 0% conserved.

### 1.4 Biodiversity Benefits Index (BBI)

The BBI (Oliver and Parkes 2003) is decomposed into a set of contributing components:

*BBI = Biodiversity Significance Score x Land Use Change Impact Score x ha*
*= $(CS\ t_o + LC)\ VC\ t_o / 200 \times ((CS\ t_n - CS\ t_o) + (VC\ t_n - VC\ t_o)) / 2 \times ha$*

Where: $CS\ t_0$ = Initial Conservation Significance
$CS\ t_n$ = Potential Conservation Significance
LC = Landscape Context
$VC\ t_0$ = Current Vegetation Condition; that is, before land use change
$VC\ t_n$ = Potential Vegetation Condition after land use change and an agreed period of time
ha = Area of land use change.

To determine the biodiversity benefits (or disbenefits) of land use change it is essential to first determine the current value or biodiversity significance of the site that will be subject to land use change. The current biodiversity value for 1 ha of the current site is scored by the *Biodiversity Significance Score* = $(CS\ t_0 + LC)\ VC\ t_0 / 200$. *Vegetation condition* is contained within this formula as a multiplier because it is largely responsible for the variation in the status of biodiversity at site scale. It is also the most sensitive to land use change. In addition, higher initial condition is also likely to be related to a higher chance of achieving successful land use change for biodiversity (greater resilience). On this basis, it therefore has a large influence on the BSS. All terms within the BSS are scored from 0–100. Division by 200 results in scores from 1–100. Once the biodiversity significance of the area has been determined the predicted magnitude and direction of change is estimated by the *Land Use Change Impact Score*
= $((CS\ t_n - CS\ t_0) + (VC\ t_n - VC\ t_0)) / 2$
A prediction of the magnitude and direction of change in the provision of habitat and other resources for indigenous plants and animals at a site scale is scored as the difference between the current and potential *Vegetation Condition*, i.e. $(VC\ t_n - VC\ t_0)$. In addition, when land use change involves the creation or loss of native vegetation it is important to score the change in vegetation type as the difference between the current and potential *Conservation Significance* score, i.e. $(CS\ t_n - CS\ t_0)$.

## 1.5 BioRisk

The BioRisk project will rely on subjective risk assessment methods described in Section 4.3. These include risk matrices (Table 1.1) and fault trees (Fig 1.3).

**Table 1.1.** At Dongolocking where the goal is "to conserve the existing biota of a landscape for 50 years", a threat analysis (extract below) was undertaken which described the probability that any one category of threat would prevent the goal being achieved, that is, will cause local extinction of a species within 50 years (Wallace et al 2003).

| **Threat** | **Method of calculating probability (p*) that threat will cause goal failure** | **p*** |
|---|---|---|
| Insufficient habitat resources and reproductive opportunities to maintain viable populations. | Most of the threats listed below will interact in a way that intensifies the problems related to insufficient habitat resources. This is taken into consideration where relevant. However, it is also possible that 'relaxation' of species at Dongolocking is not complete. There may, for example, already be populations in which deaths outnumber births, or in which recruitment has ceased. The threat of this occurring is comparatively high – estimated at 0.06 .<br><br>This threat also takes into consideration that there are some species within the Dongolocking area, such as wedge-tailed eagles (*Aquila audax*), that require very large areas to maintain viable populations, or that are nomadic or migratory. Events outside the study area will affect the local persistence of these species irrespective of management within the study area. | 0.06 |
| Introduction of new major weed | Since settlement in 1829 about 1,000 species of plants have been introduced and now grow wild in Western Australia (Hussey *et al.* 1997). Major environmentally damaging weeds have been introduced as recently as the 1990s (e.g. *Kochia scoparia* in 1990). Hobbs (1993) has calculated that | |

| | | |
|---|---|---|
| | the rate of introduction of new species has not slowed in the period 1880 to 1980. Also, many weeds have a long period persisting at a low level before they dramatically increase and become a problem. Therefore the likelihood of further, environmentally-damaging weeds being introduced or suddenly expanding within 50 years is high. The probability of a new environmental weed being introduced or expanding and causing the local extinction of a native species is calculated to be 0.05.<br><br>Assumptions:<br>• rate of introductions will remain constant or increase. There is no reason to believe that the introduction rate will decrease from that described by Hobbs (1993). In fact, given the interest in bringing new woody species and perennial grasses into the environment for production and landcare reasons, it is reasonable to assume that the introduction rate will increase, particularly as salinity increases and people attempt to bring in very robust species, and as 'miracle' species are promoted. Probability of new weeds occurring therefore considered to be 1.0.<br><br>• probability that a major environmental weed will have sufficient impact to cause the extinction of at least one native species within 50 years, either by direct competition or by habitat alteration is quite low (although high over, say, 100 to 200 years). Probability assessed to be 0.05. [Note that this estimate is conservative. Between 1947 and 1985 13 per cent of 463 exotic grasses and legumes introduced into Northern Australia became major weeds (Lonsdale 1994).] | 0.05 |
| Fire | A very large fire burnt through most of the Dongolocking area in 1927. Research is inadequate, but there is no record, to date, of two such large-scale fires affecting one location in the western wheatbelt. Therefore it is assumed unlikely to occur more frequently than once in 200 years, and that the probability of such a fire causing local extinction is 10 per cent. Therefore probability of fire causing an extinction in next 50 years is calculated as 0.005 x 50 x 0.1 = 0.025 | 0.025 |

*P = probability of a single species extinction occurring within 50 years.

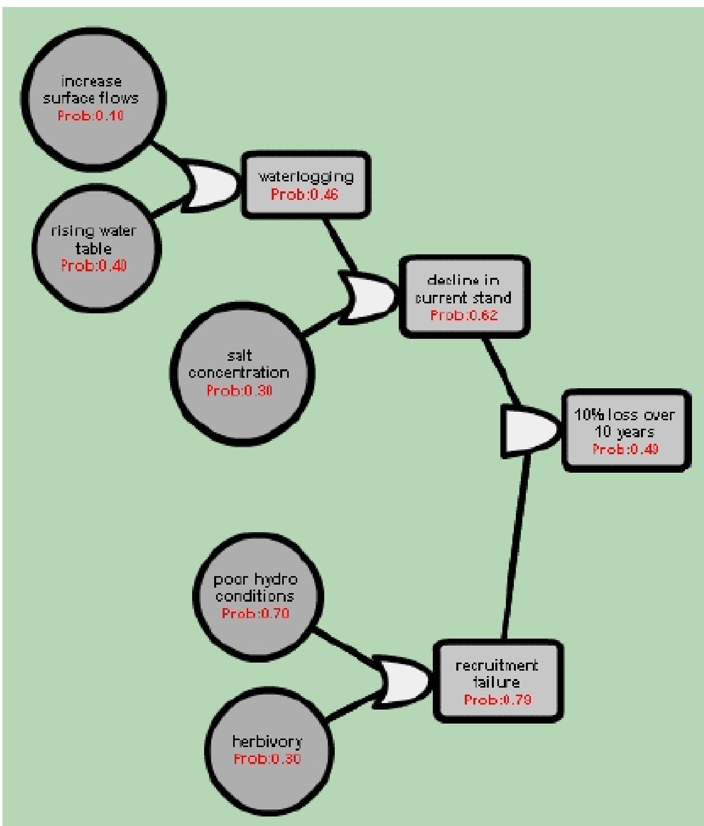


**Figure 1.3.** An example of a fault tree developed to describe pathways leading to a 10% loss in cover of *Eucalyptus camuldulensis* at Nabappie. Probabilities refer to lower bound estimates for a scenario of no management intervention (reproduced with permission of T. Walshe, University of Melbourne).