

PARKS VICTORIA TECHNICAL SERIES

NUMBER 64

# Conceptual Models for Victorian Ecosystems Pilot Program: Grasslands

Andrea K. White

August 2012

Victoria
The Place To Be

Healthy Parks
Healthy People

Parks
VICTORIA

First published 2012

Published by Parks Victoria
Level 10, 535 Bourke Street, Melbourne Victoria 3000



**Author:**

Andrea White – Parks Victoria

National Library of Australia
Cataloguing-in-publication data

Includes bibliography
ISSN 1448-4935

**Citation**

White A. L. (2012) Conceptual models for Victorian ecosystems. Pilot program: grasslands. Parks Victoria Technical Series No. 64. Parks Victoria, Melbourne.

Printed on environmentally friendly paper

**Parks Victoria Technical Paper Series No. 64**

# Ecosystem conceptual models for Victorian ecosystems

**Andrea K. White**

**This work was carried out under the Research Partners Program between Parks Victoria and the University of Melbourne**

# EXECUTIVE SUMMARY

This project is a pilot program for the development of conceptual models for each of the eight broad ecosystem groups that occur across Victoria. It is planned that for each of these systems a conceptual model will identify values, threats, processes and drivers of ecosystem health.

Natural systems are complex, with many interacting components and many potential responses to management actions. It is difficult for individuals to conceptualise these systems, and therefore to make decisions regarding their management. The information required to make informed decisions about ecosystem management is commonly fragmented and diffuse. Currently, the information required to manage parks resides in branch and regional offices, internal reports, peer-reviewed literature, unpublished data and the knowledge of experts and other external stakeholders. Ecosystem models have the potential to bring this information and knowledge together as an integrated whole, identifying threats to the biological values of the parks, the causal structure of ecosystems and the likely outcomes of specific management interventions. They will also promote understanding and support communication within PV and with external stakeholders, by providing a transparent way to communicate the rationale behind management actions. The aim is to be able to make clear, knowledgeable and explainable management decisions (McNay et al. 2006).

It was identified that an ideal modelling approach would be one that (a) effectively captures ecological interactions; (b) is simple enough for operational use; (c) communicates causal understanding effectively to managers and stakeholders; and (d) is not prohibitively expensive in the time and resources required for model construction. The Bayesian network approach is probably the method best suited to achieve the first three of these aims. However, this method is prohibitive in the amount of time required to sufficiently parameterize even a moderately complex network. A good compromise would be the use of a causal map as the comprehensive and overarching framework for each ecosystem group, and the development of State-Transition models that include management alternatives, as part of the model hierarchy. Bayesian networks would remain a possibility for use on specific management issues, where the management problem is complex and their may be diverse understandings of causality.

Addressing objectives associated with the management of natural systems cannot be restricted by incomplete or biased empirical information (McNay et al. 2006). Decisions about management will be made by managers even when faced with uncertainty. The aim of this report was to investigate methods for using the information available (from all sources) to make clear, explainable management decisions, and identify areas for further research.

# CONTENTS


**EXECUTIVE SUMMARY** ..... **ii**
**CONTENTS** ..... **iii**
**1. INTRODUCTION** ..... **1**
1.1 Objectives ..... 3
**2. BACKGROUND: GRASSLANDS** ..... **4**
2.1 Fire ..... 6
2.2 Grazing ..... 7
2.3 Weed invasion ..... 8
2.4 Soil ..... 9
2.5 Fauna & threatened species ..... 9
2.6 Management interventions ..... 11
**3. METHODS & RESULTS** ..... **13**
3.1 Ecological Models ..... 14
3.1.1 Indicators ..... 15
3.2 Causal Maps ..... 16
3.2.1 Grassland causal map ..... 18
3.2.2 Suitability and application ..... 20
3.3 Fuzzy Cognitive Maps ..... 20
3.3.1 Yanakie Isthmus example ..... 22
3.3.2 Grassland fuzzy cognitive map ..... 24
3.3.3 Suitability and application ..... 27
3.4 Bayesian Networks ..... 28
3.4.1 Grassland Bayesian Network ..... 31
3.4.2 Suitability and application ..... 32
3.5 State-Transition Models ..... 34
3.5.1 Grassland State-Transition Model ..... 36
3.5.2 Suitability and application ..... 39
3.6 Feedback regarding applicability of models ..... 42
3.7 Time lines for modeling stages ..... 49
3.8 Software ..... 52
**4. DISCUSSION** ..... **53**
**5. CONCLUSIONS** ..... **55**
**6. ACKNOWLEDGEMENTS** ..... **57**
**7. REFERENCES** ..... **58**

# 1. INTRODUCTION

This project is a pilot program for the development of conceptual models for each of the nine broad ecosystem groups that occur across Victoria. It is planned that for each of these systems a conceptual model will identify values, threats, processes and drivers of ecosystem health. The models will reflect our current knowledge of the systems and incorporate up-to-date research. The important values and processes that drive the system, as well as the things that threaten these processes or the specific values of the system will be identified. We aim for the models to effectively capture current knowledge, to be comprehensive, to gather and consolidate information and to identify knowledge gaps. The end result can be used to present contemporary understanding to managers and stakeholders as a means of decision support and communication, in a manner that is understandable and explicit. Transparency regarding the information sources and assumptions will provide a framework for assessing relative differences among alternative perspectives on the merit or demerit of future management policies (McNay et al. 2006). The models will be fully documented so that sources of information can be identified and the ideas, logic and reasoning behind the models interrogated. The ultimate aim is to be able to make clear, knowledgeable and explainable management decisions (McNay et al. 2006).

Parks Victoria has two programs which could benefit from the development of ecosystem conceptual models, these are (1) Levels of Protection (LoP), which aims to identify priorities for management and (2) the Signs of Healthy Parks program (SHP), which aims to assess performance of that management. Both programs need ways of identifying what the main values, threats and emerging issues are in these systems and the efficacy of management activities, while ensuring that different programs are consistent with one another. Conceptual models have the potential to fulfil this role, and contribute to the selection of potential variables for monitoring.

There is no single natural scale at which ecological patterns should be characterised (Levin 1992). This project focuses on the meso- or topo-scale (a few hundreds of kilometres). At this scale the main drivers are climate, rock type and topography, which together determine variations in nutrient availability and hydrology. For the broad terrestrial ecosystem types considered here environmental (e.g. soil properties, slope, landscape position) and climatic characteristics have produced recognizable and characteristic ecosystem types. These differences confer differences in the structure and composition of plant communities. A plant community is a basic unit for vegetation mapping and management, it is a relatively homogeneous plant assemblage that occurs in a specific place/time, and can be defined at a scale relevant to a land manager. The range of plant communities in an area are observable and measurable and can be linked to the processes that embody the remaining components of the system. In this project the main terrestrial ecosystem types are defined by the dominant structural vegetation type present. The coarse resolution of this typology implicitly assumes that management decisions are more or less insensitive to consideration of more detailed biotic entities. Although a practical starting point for managers and planners, this assumption is unlikely to be reasonable in all circumstances. Generalisations should be carefully assessed for each ecosystem (and possible subgroups within each ecosystem).

Sub-groups may occur within ecosystem types. For example, the dry forests and woodlands ecosystem contains a variety of sub-groups such as box-ironbark forests, semi-arid pine-belar woodlands of the north west and coastal manna gum woodlands. While a very broad model might cover all these woodlands, a model which focuses on these sub-groups and their specific values and issues will have greater predictive capacity and may therefore be more useful. These benefits need to be considered alongside the costs of formal capture of dispersed and sometimes conflicting knowledge in a modelling exercise. A decision whether these models will be required will be undertaken following the preparation of the pilot ecosystem model.

At a finer scale (finer than considered here) micro-habitats determine where individual organisms are distributed, and we generally know more about processes at this scale. One of the largest challenges in land management is finding ways of drawing together the detailed site-specific information and data, and apply them across larger regions (Bestelmeyer et al. 2003). This hinges on being able to generalize about the importance of particular processes in different ecological site types and at different scales. In some cases we find that the same types of processes are used to explain transitions (changes in state or condition) in similar ecological sites. For example, conditions in lowland sites may be influenced by changes in hydrology, surface soil structure and chemistry in relation to soil infiltration; in highlands the main influences may be erosion and loss of soil fertility. The important factor is that some subset of common processes in various combinations seem to explain vegetation dynamics within different ecosystem types, and the transitions that may occur.

This report explores four alternative approaches to modelling that may provide effective decision-support to Parks Victoria. An ideal modelling approach would be one that (a) effectively captures ecological interactions; (b) is simple enough for operational use; (c) communicates causal understanding effectively to managers and stakeholders; and (d) is not prohibitively expensive in the time and resources required for model construction. These themes form the basis for an appraisal of the merit of alternative approaches using Grasslands as a case study.

## 1.1. Objectives

The development of these models will assist Parks Victoria with key management questions: what to manage (prioritizing threats), what to monitor and what research to do. The conceptual models will provide a transparent means with which to communicate the values, threats and drivers in the ecosystem types, make explicit what it is we aim to protect, and what we measure in order to gauge our effectiveness. It is a method which can be used to convey to staff and other stakeholders of how monitoring is targeting various ecosystem processes, threats and the impacts of management actions.

The Grassland case study will establish a process for developing models for all of the other broad ecosystem types: Alps; Coastal (including intertidal, shores and estuaries); Marine (including subtidal reefs, seagrass, soft sediments and pelagic); Dry forests and woodlands; Heathlands; Inland waters and wetlands; Mallee; and Wet forest and rainforest. We expect that these models will be useful for corporate and business plans (funding cases) and also to inform policy, especially in a strategic sense.

The objectives of this report are:

- To gather information necessary to describe Victorian grassland ecosystems, including the values to be protected, threatening processes and potential management interventions.
- To use Causal maps, Fuzzy Cognitive Maps, Bayesian Networks and State-Transition models to model these systems.
- To document the strengths and weaknesses of each modeling method for use in the management of Victorian parks by PV.

# 2. BACKGROUND: GRASSLANDS

Victoria has nine broad Natural Ecosystem Groups (Figure 2.1). These groupings were derived by amalgamating broadly similar EVCs based on the 1997 Victorian Biodiversity Strategy. Basic components of the nine ecosystem types were identified in a workshop undertaken by the research branch. This work been used as a basis for further investigations into grassland systems, including a review of the literature, an elicitation process with experts and other stakeholders and ongoing involvement and consultation with Parks Victoria research, conservation, and on-ground staff.

There are two steps in this process, outlined as follows:

1. Gather and consolidate of information and knowledge of pilot system (grasslands), and explore alternative approaches to gathering information for models.
2. Explore of a number of different modeling methods, documenting the strengths and weaknesses of each approach in terms of their applicability to PV purposes. Methods will include: Causal maps, Bayesian Networks, Fuzzy Cognitive Maps and State-Transition models.

Lowland grasslands in south-eastern Australia have been greatly depleted in area and condition throughout their original range, including in Victoria (Cole and Lunt 2005, Verrier and Kirkpatrick 2005). Drivers of change have been clearing, livestock grazing and cultivation (Prober and Thiele 2005). Floristic changes included a rapid breakdown of the grass tussock sward in *Themeda*-dominated grasslands, which was then replaced by secondary native perennial grasses, and an increasing abundance of exotic annuals. The trends in Wallaby/Spear grass communities are less well understood, but included a reduced diversity of native grasses, a decline in many perennial forbs, and an increase in exotic and native annuals (Prober and Thiele 2004).

Since European settlement, the trends in temperate grazing lands have included (1) a decline in diversity of native perennial herbs, (2) a shift towards species with cool-season growth and (3) a dominance of mostly exotic annuals (Dorrough et al. 2004). An increase in watering points and supplemental feeding in droughts has allowed a high density of native and introduced herbivores. The foot pressure of introduced herbivores is high and has caused damage to the structure of the soil surface (Mussared 1997). Grazing has also resulted in an increase in bare ground, especially at the end of summer when soil moisture is low.

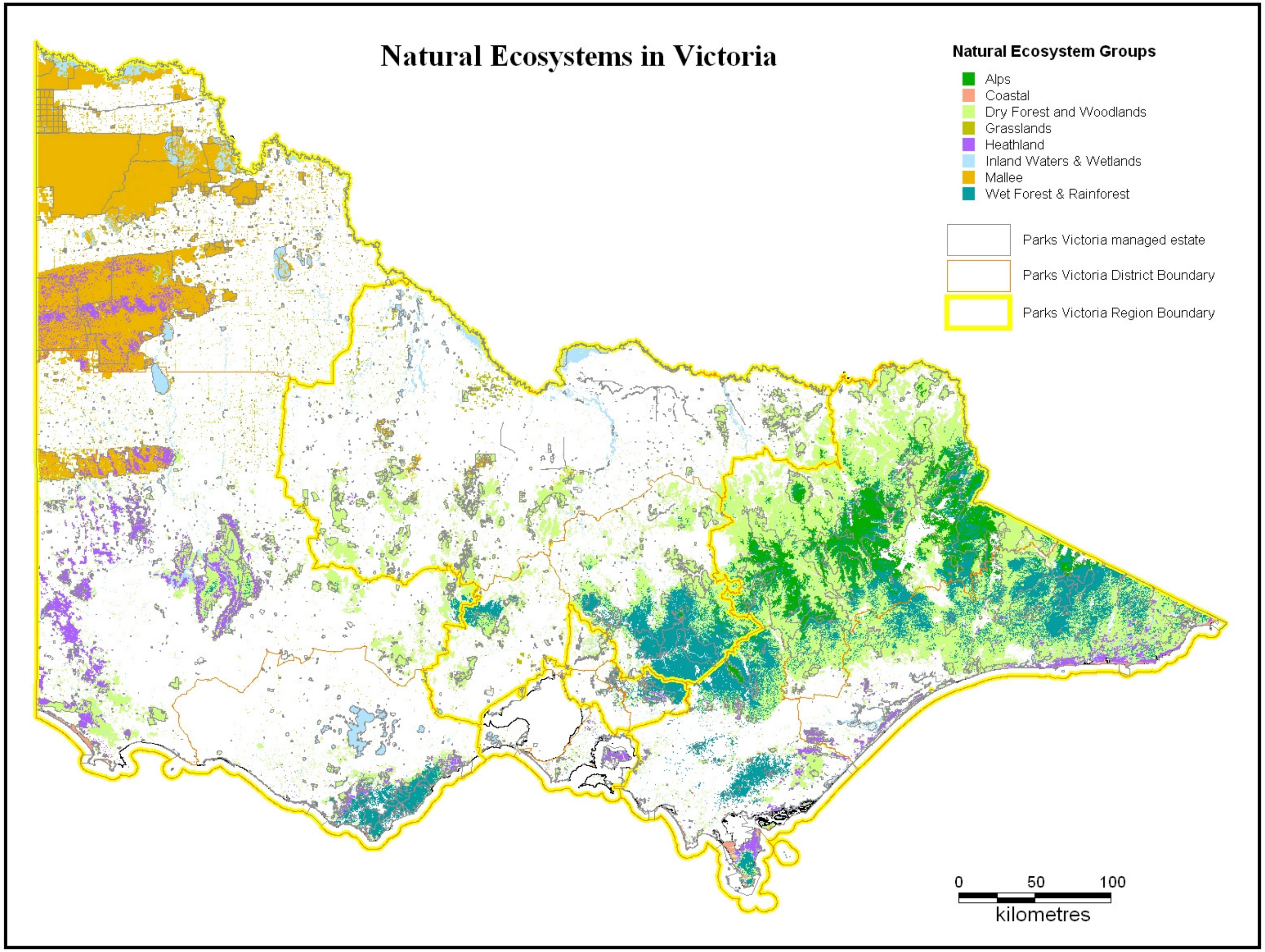


**Figure 2.1:** Natural ecosystem groups in Victoria, these were derived by amalgamating broadly similar EVCs based on the 1997 Victorian Biodiversity Strategy.

Gap formation in late summer and early spring is a key precursor to invasion by exotic annuals (as they germinate at this time and grow throughout winter), and declines in native perennial species (Dorrough et al. 2004). The control of grazing in late summer is also important as the soils are dry and easily degraded. Selective grazing by cattle results in the dominance of unpalatable species, and gradually reduces the productivity of the grassland as a resource (Dorrough et al. 2004). Widespread clearing and loss of deep-rooted perennial grasses leads to disruption of physical processes such as modification to microclimates, soil nutrient flows and hydrology. These changes may lead to salinization and threaten remaining flora and fauna (Prober and Thiele 2005).

Dorrough et al. (2004) categorised Australian temperate grasslands into two groups, as follows:

1. Fragile systems that have low resource availability, where water or nutrients limit plant growth and persistence. Grazing in these systems is more likely to result in soil degradation, and extreme microclimates.
2. Robust systems with high resource availability. In these systems grazing sensitive natives tend to be replaced with productive or invasive species; however grazing may perform a useful function (the removal of biomass, see below).

Drier grasslands occur in areas where the annual rainfall is <550mm; these grasslands are dominated by wallaby, spear and windmill grasses, and probably a denser shrub component (Prober and Thiele 2005). Grasslands in wetter areas are dominated by *Themeda triandra* (referred to as '*Themeda*'). *Themeda* is the dominant species of natural grasslands on volcanic soils in southern Victoria (Morgan and Lunt 1999). It is a C4 perennial tussock grass and grows mostly from the months of October to January. *Themeda* seed set occurs mid summer, germination in spring and it is most productive in the first 3-4 years after fire, after which productivity declines (Ibid.). Post-fire recovery seems slower in swards that have not been burnt regularly. *Themeda* tussocks are fire resistant, and tillers resprout after summer fires from protected basal apices. The species has no means of vegetative spread (Ibid.). When the sward remains undisturbed aerial tillers may form, although these are easily damaged by trampling.

## 2.1. Fire

Before European settlement, and the clearing and cultivation of grassland habitat, fire was a regular source of disturbance for grasslands. It is estimated by Morgan and Lunt (1999) that intervals of less than five years are necessary to maintain the health of these *Themeda*-dominated systems. If the inter-fire interval exceeds six years the number of shoots that sprout from the base of the tussock (tillers), and the total number of tussocks, begin to decline. Morgan and Lunt (1999) found that if the inter-fire interval reached 11 years few live tillers or tussocks remained and that below-ground biomass had been reduced. They also found that with increasing time since fire the canopy of live leaves would become elevated above the soil surface, dead leaves accumulated around and over the tussock base and productivity declined. With no disturbance for 11 years the canopy of the sward began to collapse and form a thick layer of dead material over the soil surface. Morgan and Lunt (1999) noted that a single fire 12 years after last burn did not immediately return the grassland to a good state.

Burning is also important for inter-tussock native flora that can be rapidly eliminated from grasslands due to severe competition from *Themeda* in the absence of disturbance. The decline in floral diversity may be irreversible given that the seed bank for many of these species is transient. These species are unlikely to return without management intervention (Prober and Thiele 2005). Faunal species, such as the Earless Dragon and Plains Wanderer, also rely on the creation of inter-tussock spaces (Section 2.5). Burning in patches is important as it minimizes the potential for soil erosion, and also provides areas to act as refugia for faunal species (Prober and Thiele 2005).

## 2.2. Grazing

The vulnerability of a system to the impacts of grazing depends on the productivity of the system. Species diversity may be promoted in productive systems and reduced in unproductive systems (Lunt et al. 2007). Australian ecosystems evolved with low grazing pressure from native herbivores, and because of this large herds of sheep and cattle that were introduced post-settlement had immediate and severe impacts on soils, landscape processes and flora and fauna (Mack and Thompson 1982, Lunt et al. 2007).

Grazing influences grassland condition via a number of pathways: there are floristic changes related to differences in palatability and defoliation tolerance; soil compaction and disturbance; weed invasion; modifications to nutrient cycling and a loss of structural diversity, which may lead to further declines in flora and fauna (Dorrough et al. 2004, Lunt et al. 2007). Livestock also cause soil compaction and erosion, pugging and nutrient deposition. Heavy continuous stock grazing causes a shift from C4 to C3 grasses, from perennials to annuals, and from native to exotic species. Extremely high grazing pressures from high numbers of native grazers can also cause serious ecological degradation (Grigg 2002).

*Status quo* management, as defined by Diez and Foreman (1997), is based on the premise that past grazing management has produced the current suite of species in the grassy ecosystem, and that if the same grazing regime is continued that these species will persist. This approach makes a number of assumptions including that current grazing-grassland-climate system is stable and that the current levels of stress are not causing changes in ecosystem function (Dorrough et al. 2004). It also does not take into account that species may persist in the short-medium term, but be slowly reducing in abundance. Grazing and/or grazing removal may have positive, negative or neutral impact on the diversity and composition of native plants, and predicting which one of these impacts will be most likely is difficult. Factors that may influence the outcome include the following (Lunt et al. 2007):

- Soil and ecosystem processes,
- Site productivity,
- Relative palatability of dominant species,
- Species specific factors influencing plant recruitment, and
- Spatial scale and landscape context.

Grazing may be a useful management tool if it controls biomass of existing potentially dominant grazing-sensitive plants (native or exotic); prevents the encroachment by undesirable grazing-sensitive plants; provides disturbance niches required by rare or significant plant species; maintains habitat structure; and/or enhances diversity of species and vegetation structures (Lunt et al. 2007).

When a grassland is grazed the structure of the sward changes in a patchy manner, there is a mosaic of short and tall patches which provides a diversity of habitats. However, as grazing intensifies the area of short-grazed patch increases until the paddock has a short, even appearance (McIntyre and Tongway 2005). Heavy grazing can affect soil and water function and ultimately cause erosion.

## 2.3. Weed invasion

Soil nitrogen limits plant growth in most grassy ecosystems worldwide (Prober and Thiele 2005). In Australia, available nitrogen increases with soil disturbance and addition of nutrients, which encourages growth of nitrogen-loving annual exotics. This produces a seasonal spike in nitrogen through breakdown of plant material after plants die each year, and in turn favours ongoing persistence of annuals over native species. The seasonal peak in soil nitrate may be suppressed by the addition of carbon to the soil (e.g. using sugar, saw dust), this increases the amount of carbon in relation to nitrogen, allowing soil microbe abundance to increase and which uses up available nitrogen (Prober et al. 2002, Prober et al. 2005). The re-introduction of *Themeda* will also reduce oil nitrogen.

As the abundance of native plant species is reduced the soil seed bank also becomes diminished. This can occur rapidly as the seeds of most native grassland species are short-lived. Through this process the seed bank may become dominated by exotic annuals. In addition, grazing has the effect of producing bare ground, which also encourages weed invasion. The timing of disturbance is also very important, for example, burning can be timed to maximize the negative effect on weed species and maximize the positive effects on native flora and fauna (Prober et al. 2004). One of the serious and relatively new invasive species is *Nassella neesiana* (Chilean needle grass); a healthy *Themeda* sward may play a role in resisting invasion of such species (Morgan and Lunt 1999).

## 2.4. Soil

Heavy grazing can affect soil and water function and ultimately cause erosion. McIntyre and Tongway (2005) found that water infiltration and nutrient cycling indices declined as grazing pressure increased. They also found that the stability index for soils (estimated using amount of bare ground, litter cover, vegetation cover, erosion, surface resistance to disturbance, etc) was reduced in the most heavily grazed plots.

Soil nitrate rises to high levels over summer and autumn encouraging lush growth of exotic annuals as they germinate in autumn. Suppressing this nitrate peak may reduce vigour of annual exotics and enhance the establishment and competitiveness of native species. This can be achieved by adding carbon to the soil (e.g. as sugar or saw dust). This should have the effect of increasing the carbon-nitrogen ratio, causing soil microbes to increase in number and use up available nitrogen (Prober and Thiele 2005).

McIntyre and Tongway (2005) found the following trends as grazing pressure increased:

- A decrease in infiltration,
- A decline in nutrient cycling, and
- Reduced stability of soil surface.

## 2.5. Fauna & threatened species

The native faunal assemblages of grasslands have been very much depleted and simplified since settlement; of the medium sized ground-dwelling mammals that once inhabited these areas most are now extinct, 25% of woodland birds are in serious decline and foxes and rabbits have become widespread (Prober and Thiele 2005). The fragmentation of habitat from clearing is central to the ongoing faunal decline (disrupted dispersal, inadequate habitat size, discontinuity of seasonal food supply, reduced gene flow, reduced genetic diversity, etc). Historical records indicate that medium-sized ground-dwelling mammals were once abundant and diverse, this probably influenced ecological function (Prober and Thiele 2005).

Tussock grasses protect the soil surface and provide habitat for birds, reptiles and invertebrates. Other species rely on the gaps between the tussocks and their diversity is particularly high where the thick thatch of *Themeda* is periodically removed by fire (Prober and Thiele 2005). The impact on invertebrate communities of grazing is pronounced, with more intensive grazing systems having lower invertebrate diversity and abundances (e.g. Rushton et al. 1989, Yen 1992, Abensperg-Traun et al. 1995). The larval stages of number of native moth and butterfly species feed on native grasses, and are adversely affected by grazing, fertilizer application, and the sowing of exotic pastures (Neyland 1993 in Dorrough et al. 2004).

Grazing causes a loss of litter cover and degradation of the surface soil structure and microtopography. High grazing pressure also reduces litter and biomass and creates an even sward structure, which is detrimental to faunal species such as the Earless Dragon and Plains Wanderer, which have a preference for an open sward (Dorrough et al. 2004). In contrast, the Striped Legless Lizard prefers the habitat provided by a closed sward. These contrasting habitat preferences suggest the importance of intermediate spatial and temporal variability in density and biomass of the grassland habitat.

A number of threatened species rely on grassland habitats, and include the following flora and fauna:

- Brittle Greenhood (*Pterostylis truncate*)
- Basalt Sun-orchid (*Thelymitra gregaria*)
- Basalt Greenhood (*Pterostylis basaltica*)
- Adamsons Blown-grass (*Agrostis adamsonii*)
- Golden Sun Moth (*Synemon plana*)
- Striped Legless Lizard (*Delma impar*)
- Southern Lined Earless Dragon (*Typpanocryptis lineata lineate*)
- Corangamite Water Skink (*Eulamprus tympanum marnieae*)
- Plains Wanderer (*Pedionomus torquatus*)
- Curly Sedge (*Carex tasmanica*)
- Button Wrinklewort (*Rutidosis leptorrhynchoides*)

## 2.6. Management interventions

Potential management interventions in grasslands include manipulations of grazing pressure (native and introduced) and fire regimes, mowing or slashing, selective use of herbicides, carbon addition, and the re-introduction of native species. A particular management intervention will target specific issues in a grassland (or a section of grassland). For example, it may focus on weed control, biomass management, maintaining habitat requirements for specific species, improvement of soil structure or nutrient balance. Any one management action may also have multiple benefits.

Degraded remnants may have a large seed bank of annual exotics, which need to be reduced in abundance to make replacement with native species seeds possible. Scalping, which involves the removal of a thin layer of top soil, may be used to remove the weed seed bank (Prober and Thiele 2005). This may also influence soil nutrient levels. Strategic burning may be used to manage weed abundance. Cool-season annual grasses (an important group of exotics) can be reduced by burning in spring before established plants set seed. Spring burning may require use of knock-down herbicide or steam to reduce fuel moisture content to flammable levels (Prober and Thiele 2005). It may be most effective on sites with few broad-leaf annual exotics, as these usually have longer-lived seed banks and can increase on bare soil.

Pulse (short duration and high intensity) grazing may be used to control weeds (Dorrough et al 2004, Prober and Thiele 2005), especially where burning is not possible. Herbicides and repeated slashing may also be used. Sward re-establishment is important as a dense stand of native perennials will out-compete weeds and maintain low levels of available nitrogen and phosphorus (Prober and Thiele 2005). No ecological methods are yet known for serious invasive perennials such as St. John's Wort as they are similar to native perennials in their ecology (Ibid.). These species should be dealt with promptly with herbicide or manual removal. Avoiding disturbances that encourage their invasion is one important preventative measure.

Verrier and Kirkpatrick (2005) found that mowing was superior to moderate grazing in conservation outcomes in that it resulted in greater cover of rare or threatened species, greater native cover and less exotic grass cover. They also found that the removal of slash would seem to be beneficial in that it reduces the nutrient status of the ecosystem – an outcome beneficial to reducing weed infestations. They noted that it may be necessary to retain an unmown area to supply seeds. They concluded that frequent burning was not necessary to maintain good condition in the medium term where mowing or moderate grazing were present (Verrier and Kirkpatrick 2005).

Native plant species that have been lost from a site are not likely to come back without management intervention as their distribution in the landscape is too sparse to rely on natural seed dispersal (Prober and Thiele 2005). Failure of many forbs and grass species to spread into restored *Themeda* grasslands may be due to one or more of the following factors:

- Competition with exotics,
- Soil nutrient enrichment,
- Weed seed banks, and/or
- Establishment conditions (e.g. high nitrogen).

Pulse grazing in mid summer may be used to reduce the competitive effects of the dominant introduced grasses and provide summer growing native grasses enough time to grow and fill gaps prior to autumn germination of annual exotics. There are major advantages to burning in patches, these include the minimization of soil erosion, and providing faunal refugia.

# 3. METHODS & RESULTS

The preceding section's description of Grassland dynamics represents a written synthesis of the research literature, together with commentary on ecological values, threats and possible management options. Literature reviews are by far the most common approach used in evidence-based decision-making. However, it's unlikely to be the best approach. The complexity of interactions, the varying scale involved in individual studies, and the speculative narratives of cause and effect linking management actions to outcomes all conspire against the reader's ability to form a coherent understanding.

Graphical models provide a more effective approach. Axelrod (1976) contended that "when a cognitive map is pictured in graph form it is then relatively easy to see how each of the concepts and causal relationships relate to each other, and to see the overall structure of the whole set of portrayed assertions". de Bruin et al. (in press) tested understanding of medical risks among study participants that were provided communication materials based on written scenarios or graphical models. Graphic representations substantially outperformed scenarios in improving people's understanding of risks. This section explores four alternative approaches to graphical capture of understanding of Grassland dynamics.

The first step in the modeling process involved the mapping of a (conceptual) causal map for grasslands. This was carried out using specialist software (CmapTools, IHMC 2008) which captured causal information and represented it in a map. These maps have the capability to convey relatively complex ecological information to broad audiences. They also have the capacity to structure the causal model in a hierarchical manner, so that different levels of detail in specific areas can be easily accessed. This allows stakeholders to apply their knowledge to their specialist area to an appropriate level of detail. Photos, maps, documents, references and other information can be also attached to relevant parts of the model to help convey our collective understanding of the system. This stage of the modelling process focused primarily on getting the structure of the model to represent, as clearly and concisely as possible, the relationships between threats and the values and processes we are trying to protect.

The next step in the process involved a more formal parameterization of the model, where the relationships between threats, values and processes were informed by different types of data. This may involve quantitative or qualitative data, anecdotal evidence, expert opinion, and the output of other modelling processes. The linkages and interactions between values, threats and processes were described numerically to indicate their strength and importance. Three types of model that involve numerical extensions of causal maps were explored: Fuzzy Cognitive Maps (using Hot Fuzz, CSSE 2008), Bayesian Networks (using Netica, Norsys 2005) and State-Transition models.

## 3.1. Ecological Models

Ecological models are used to examine, compare and contrast hypotheses that can explain observed patterns in natural systems. An individual model that is coherent and consistent with observations can be thought of as a formal hypothesis of system dynamics (Neuhauser 2001). Statistical modelling tools in ecology have traditionally been based on frequentist methods (Pollino et al. 2007). These have been used explain patterns in ecological systems where causes are single and separable, and discrimination can be provided using pair-wise hypotheses and a simple yes or no answer (Holling and Allen 2002). However, causes in ecological systems are likely to be multiple, overlapping (Holling and Allen 2002), and data are usually sparse.

Natural systems may be exceedingly complex, with many interacting components and many potential outcomes from management actions. It is difficult for managers and individual domain experts to conceptualise these systems, and therefore to make decisions regarding their management. Models of these systems may also be large and complex. However, diagrammatical network-based models are modular, which allows the full model to be separated into smaller, more manageable parts which can be developed separately and then re-aggregated (Pearl 2000). These models will help to identify threats and decision alternatives, determine the likely outcomes from specific management interventions and how components of the system are likely to interact.

Predicting ecosystem behaviour is inherently uncertain, and knowledge of these systems will always be incomplete. In addition to this the system itself is dynamic and evolving due to management interventions and other anthropogenic impacts (Walters and Holling 1990). Levins (1966) proposed an approach whereby models are used to simplify in a way that ‘preserves the essential features of the problem’. He pointed out that all models leave out a lot of information and that they are false, incomplete and inadequate. Levins (1966) suggested precision could be sacrificed for realism and generality using flexible (often graphical) models, that assume that relationships are increasing or decreasing, convex or concave or greater than or less than a particular value, instead of specifying the mathematical form of an equation.

Ecosystems occur at scales that are not generally amenable to manipulative experiments, and the uncertainty associated with extrapolation from smaller scale experiments is difficult to quantify (Stow et al. 2003). However, management actions can act as large-scale experiments and will enhance what is learnt through further research (Stow et al. 2003). This approach, termed adaptive management, is where the consequences of management are monitored so that management efficacy can be gauged and actions modified in response to feedback. Adaptive management requires the development of conceptual models that can outline the likely consequences of management interventions, and the important threats and processes that may be involved.

The different modeling methods present in the following sections (3.2-3.5) differ in their ability to carry out inference (i.e. provide answers to 'what-if' scenarios). Inference is important for decision making as it supports prediction based on specific management interventions, and provides a framework for decision-making for individuals other than the experts used to construct the model (Nadkarni and Shenoy 2001). Modeling methods differ in the clarity of inferences. Some (e.g. Bayesian Networks) explicitly state the probability of specified outcomes. Others (e.g. causal maps and Fuzzy Cognitive Maps) simply explore the potential effects of a management decisions in vague qualitative terms (Eden 2004). Casual maps can be used as a starting point for both Fuzzy Cognitive Maps and Bayesian Networks. The state-transition models presented in Section 3.5 are underpinned by a causal model as well but are more heavily focussed on the alternative management actions and their consequences.

### 3.1.1. Indicators

Inferencing in a graphical model focuses on the response of a query node under various what-if scenarios. Query nodes should be good indicators of broader ecological condition. Thresholds of probable concern can be used, described by a range of spatially and temporally bounded indicators of the system's response to the main potential agents of change (Rogers and Biggs 1999). An operational definition of the desired system condition that reflects scientific rigour and broader societal value systems are needed. These thresholds can be monitored and represent statements or hypotheses of the limits of acceptable change in ecosystem structure, function and composition.

For ecological models to be useful in decision support they must provide a predictive link between management actions and ecosystem response; in addition to this the decision support tool will be more effective if the ecosystem response is represented by an attribute that stakeholders care about (Reckhow 1999, Borsuk et al. 2004). Suter (1993) lists three desirable attributes of an indicator: (a) ecological importance, (b) social relevance, and (c) ease of measurement. Trade-offs among these three attributes are typically required. The example used by Borsuk et al (2004) involved process-based biophysical models that allow the prediction of water quality characteristics, such as dissolved oxygen concentration, at a fine spatial and temporal scale. While these variables are useful indicators of water quality and are easily measured, they have little meaning to decision-makers and the general public, who are likely to be more interested in things such as harmful algal blooms and fish kills.

Attributes used to gauge the condition of grasslands have included considerations of the cover and abundance of indigenous and introduced flora and fauna, plant structural diversity, soil structure and stability, productivity, levels of biomass/inter-tussock gaps and intact, self-sustaining ecosystem processes.

Indigenous plant community composition and distribution (including consideration of important functional and structural groups), is related to the theorised pre-settlement or historic state. Prober and Thiele (2005) proposed that a pre-disturbance grassland would have no weeds and high diversity and cover of native species. Therefore, a grassland in good condition would also have these attributes. The level of biomass and time since last fire (or other disturbance) is also important in *Themeda*-dominated systems, which require periodic removal of plant material (Morgan and Lunt 1999).

A heterogeneous grassland structure is considered to be a positive attribute, as it provides the niches required by rare or significant species and enhances diversity of flora and fauna species (Lunt et al. 2007). High structural diversity occurs in grasslands when disturbance is patchy, creating a spatial and temporal mosaic of different species and life forms. A decline in structural diversity leads to declines in flora and fauna (Prober and Thiele 2005).

Soil stability is estimated using the amount of bare ground, litter cover, vegetation cover, soil surface resistance to erosion, compaction, water flow patterns and erosional structures ((McIntyre and Tongway 2005, Pellant et al. 2005). A grassland that is self-sustaining has natural cycles intact, this includes nutrient cycling, hydrology (including low levels of soil compaction and erosion and high infiltration), seed supplies, high floristic diversity and structural diversity at a variety of scales (Eddy 2002). Individual species mortality and reproductive success must also be considered (Pellant et al. 2005).

## 3.2. Causal Maps

### Overview

Causal maps (also known as influence diagrams) are a type of network-based model that is used to represent the domain knowledge of experts (Nadkarni and Shenoy 2001); they express the judgement that certain events or actions will lead to particular outcomes. The components of a causal map are the factors that influence the system being modelled (nodes) and the causal relationships between the nodes (arcs or arrows). The direction of the arrows imply causality. The software used in this project (CmapTools) allows for explanatory text to be included as part of the arrow and helps to describe the nature of the relationship.

### Strengths

Links in causal maps indicate causation; most people are able to express their understanding of a system in this manner (Cain 2001). Of the four approaches explored in this report, causal maps are the most accessible to those without formal training in modelling. Individuals reason by accumulating possibly significant pieces of information and organising them in relation to each other and combine them in order to make conclusions and decisions. We use such processes to put together cause and effect events into series to predict the future course of events (Nadkarni and Shenoy 2001).

Network based models, such as causal maps, represent knowledge more descriptively than other types of models (e.g. regression analysis) and because of this they are particularly useful in decision analysis (Nadkarni and Shenoy 2001). They have been used extensively in policy analysis (Axelrod 1976) and management sciences (e.g. Klein and Cooper 1982) to represent factors that influence decision making. Causal maps describe different domains of knowledge but also identify how they are linked; they are used in the formulation of problems and hypotheses, and to explore the potential effects of management decisions (Eden 2004).

Casual maps are well suited to ecological problems, where knowledge is often imprecise, there may be diverse understandings of causality and the impacts of intervention, and very often a need to develop a common understanding amongst stakeholders (Hobbs et al. 2002). These models can be used to promote communication and understanding between participatory stakeholders or experts, and to simulate different management scenarios to determine an optimal set of management actions. Different perceptions of causality may be revealed, providing an opportunity for learning and consensus to occur. They can form an accessible knowledge repository and a medium for communication (Mingers and Rosenhead 2004).

### Weaknesses

Causal maps are not well suited to inference. The net influence of multiple causal pathways is typically indeterminate. Causal maps do not model uncertainty (all variables in the maps have the same level of certainty) and the representation of the decision variables is static (Huff 1990, Laukkanen 1996). Identifying the level of uncertainty is important in making inferences because observations of variables may be uncertain, information may be incomplete, or the variables involved may be vague. Causal maps can not easily incorporate threshold effects.

Causal maps also do not depict how beliefs of decision-makers about some target variables change when decision-makers learn additional information about relevant situational factors or decision options represented in the map. Such a dynamic approach is important in not only drawing inferences but also in learning about causal relationships and representing complex and uncertain decisions (Heckerman 1996) (in contrast to Bayesian Networks, which have the capacity to do this, see Section 3.4).

While causal maps can be drawn free-hand, their effective use may require specialist software that is made accessible to all domain experts.

### 3.2.1. Grassland causal map

An extensive literature review (Section 2) was conducted for Victorian grasslands, and an initial causal map was produced (Figure 3.1). The red boxes represent the processes that threaten grasslands, and impact on the indicators of grassland state, which in turn effect the things that we value about these grassland systems (represented by the blue boxes), which include the conservation of threatened species and communities, and native herb, forb, invertebrate, reptile and bird species. Grassland values were sourced from the literature and from grassland experts. The yellow boxes represent what could potentially be measured in order to monitor the effectiveness of management strategies that aim to protect these values.

The most important threats to grassland persistence are represented by dark red nodes. The processes through which these act on the grassland indicators are depicted by the intermediate nodes. For example, an inappropriate grazing regime (outlined in greater detail by the state-transition model, Figure 3.5) will cause soil compaction which damages the soil crust and impacts on hydrology by reducing infiltration (Bowker et al. 2006). Grazing may also change the species composition of the grassland as some species are more palatable than others, and species vary in their tolerance to repeated defoliation. Cultivation introduced nitrogen-fixing species and phosphate fertilizer has been added to aid in the growth of pasture plants. This has conferred a competitive advantage to introduced species over many of the indigenous grassland plants. A change to the pre-settlement fire regime has also had an impact on grasslands, which require periodic burning to reduce biomass and created inter-tussock gaps for indigenous herbs and forbs (see state transition models in Section 3.5).

The symbols associated with a number of the nodes in Figure 2 indicate where extra information has been provided (e.g. state-transition models, sub-networks, photos, maps). These provide additional detail that cannot be included in the main model (see Figure 3.6 and 3.7). Certain nodes also display details of key references when the cursor is place over the node; this allows the model to be used as a repository of information that is very easily accessed.

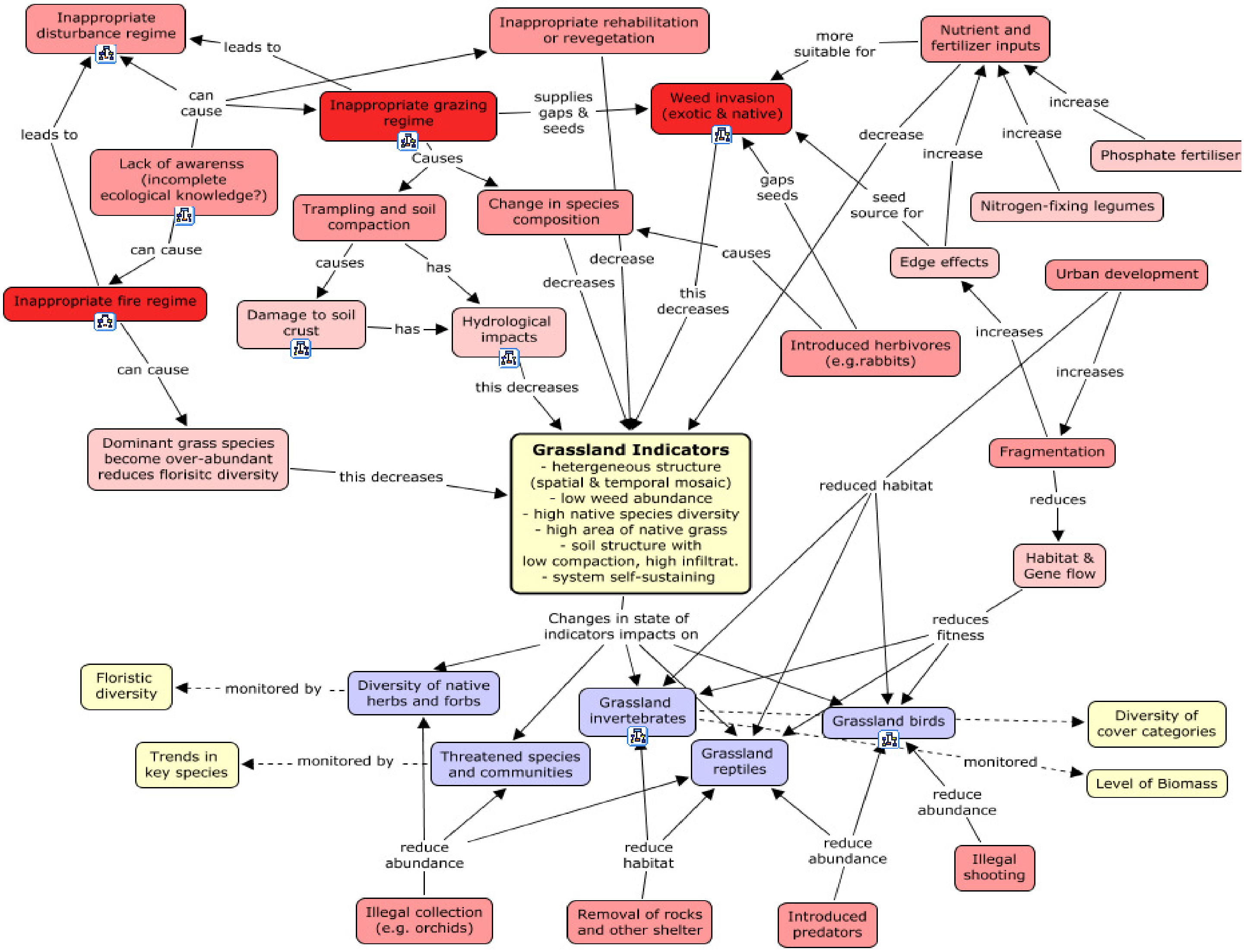


**Figure 3.1:** Causal map for *Themeda*-dominated grasslands. The red boxes represented threats, the blue boxes values and the yellow boxes features of the grassland that could be measured to monitor the trends in key values. Further details of the grassland indicators listed here can be found in Section 3.1.1.

### 3.2.2. Suitability and application

The casual map developed for grasslands was presented to a number of experts to elicit feedback. A model walk-through was conducted at a meeting with PV staff from the Research and Management Effectiveness Branch and the Healthy Parks Branch, and with experts individually. Overall feedback was positive. The ability to represent complex systems and the option to link to submodels (which contain extra detail) were perceived as useful features of this method. The difficulty in representing threshold effects was considered a drawback, though these effects can be incorporated in the more detailed models in the model hierarchy (e.g. state-transition models). The use of these models as a repository of knowledge and understanding of the ecosystems was considered an advantage, especially in having all of the relevant facts and information in the same place.

To interact with this model (i.e. to change the nodes or relationships) requires specialist software (CmapTools, http://cmap.ihmc.us/conceptmap.html). This software is free, and the developers provide a server to allow maps to be shared and constructed collaboratively. Lines of comment/suggestions/criticism are saved as threads, documenting model development and allowing individuals to work on models together from different locations. However, it does require some time to become familiar with the software, and IT systems that allow its installation.

## 3.3. Fuzzy Cognitive Maps

**Overview**

Cognitive maps are also based on the concept of causality, capturing and representing an individual's beliefs about a particular domain of knowledge in a network of causal assertions (Axelrod 1976). Cognitive maps are useful for decision support as, while decision makers know what they believe, they are not always able to make correct deductions from the full complexity of their many interrelated beliefs (Axelrod 1976).

The cognitive map is represented in the form of a directed graph composed of nodes and arcs, where the nodes represent concepts the individual has concerning a particular domain, and the arcs (arrows) between them represent the direction of causality (Axelrod 1976). Positive and negative are the most basic values a relationship can have on a cognitive map. A positively signed arrow indicates that there is a perceived positive causal link such that an increase in the cause (concept at the tail of the arrow) generates an increase the in the effect (concept at the head of the arrow). A negatively signed link indicates that changes in the cause variable will produce changes in the opposite direction in the effect variable. Weighting overcomes the problem of indeterminacy in inferencing associated with causal maps. Maps from different sources (experts) may use different concepts, different arrows, different signs, or different weight (Stylios et al. 1997).

Fuzzy Cognitive Maps (FCMs) were developed by Kosko (1986) to represent ‘hazy degrees of causality between hazy causal objects’. FCMs have been used to help to identify potential strategic options (Montibeller and Belton) or to develop plausible, internally consistent narratives for what could account for given observed changes (Axelrod 1976). Their use in ecology has been limited, although recent applications (e.g. Kok (2009), Ramsey and Norbury (2009)) suggest FCMs may be used more commonly in the future.

FCMs balance the trade-off between the burden of knowledge elicitation and the strength of inference possible from such elicitation. In general terms, fuzzier knowledge is easier to acquire but also harder to process and make inference from (Kosko 1986).

### Strengths

Among the strengths of FCMs are their modest elicitation burden and that they can accommodate feedbacks between different model variables (compared to Bayesian Network models which cannot, see Section 3.4), a common feature of ecological systems.

The graphic structure of FCMs are able to provide a lucid representation of complex systems using an approach can be quickly explained to stakeholders (Kok 2009). There is a high level of integration, connecting the different sub-models that cover different areas/concerns, an important feature as ecological models are usually a complex of different sub-models (Kok 2009). FCMs force users to be explicit about the strength of relationships between ecosystem components, producing semi-quantitative output and can be used to inform other (more quantitative) models. FCMs provide insight on effects of impacts and consequences of management actions, and underlying assumptions made by stakeholders are made explicit (Kok 2009).

The structure of FCMs allows causality to be propagated, by translating the relationships represented in the cognitive map into an equivalent adjacency matrix; making computations and qualitative inference possible (see Table 3.2). Using fuzzy causal algebra fuzzy inputs are processed as systematically as real-valued inputs, but the output can also be fuzzy (Kosko 1986). Fuzzy cognitive maps are useful in they can be used to test that system components are correctly represented (linked) in testing the changes in model simulations. They also help to clarify understanding of system while undergoing the elicitation process. This method may be useful for some targeted elicitation, perhaps focussing on a specific system feature (sub-component of the overall ecosystem model). Variations in graph metrics (part of the Hot Fuzz software) signify differences in structures that imply different versions of causality. Pegler (2009) proposed that it may be important to investigate the different hypotheses associated with these perceptions (e.g. in designing an adaptive experimental management response).

If a completely convergent map cannot be realised, either because the group negotiation procedure did not result in consensus, or because there was no negotiation process, aggregation can be done mathematically. Kosko (1988) describes a simple additive process of combining the adjacency matrices of individual FCMs of the same domain that have some overlapping and some unique properties. In this process, all positive and negative assertions on the strength of a relationship are summed together, and weightings may be applied according to the relative credibility of the expert. Different perceptions of causality may be revealed, providing an opportunity for learning and consensus to occur.

**Weaknesses**

The tool is designed to be simple and transparent, and therefore it has important drawbacks, Kok (2009) proposed that it should be viewed as a tool that can become part of a larger toolbox. The outputs of the model are semi-quantitative and therefore the strength of drivers and links can only be interpreted in relative terms (Kok 2009), cannot be formally updated with data or incorporate conditional relationships (both of these things are possible with Bayesian Networks).

In addition to this, not all factors can be included (Hot Fuzz allows 20 nodes). The role of weighting is essential but methods can be *ad hoc* (Kok 2009). The methods for semi-quantification are not very structured, and the FCM approach may increase time pressure (stakeholders usually note a lack of time during workshops).

FCMs have no capacity to address trade-offs in the importance of two or more values. Pegler (2009) used multi-criteria analysis to address value-based trade-offs in his decision support case-study. Pegler (2009) found that differences in map structure were not found to have a substantial association with the inferred effect on goal concepts (see below).

## 3.3.1. Yanakie Isthmus example

Pegler (2009) applied a FCM approach to a real-world decision problem: the management of coastal grassy woodlands at Wilsons Promontory National Park. A workshop was held over two days; the group included PV Staff (operational staff, ecologists and a fire ecologist), an ecologist from the Department of Sustainability and Environment, and an animal ecologist from the University of Melbourne.

Participants were prepared by the provision of material which included details of the problem and fundamentals of FCMs. A pool of concepts (model “nodes”) had been identified prior to the workshop from the relevant literature, and included the two management goals (cover and abundance of ground layer species and seedling abundance of Coast Banksia and Drooping She-Oak) and management actions (burning to control Coastal Tea Tree and culling of grazers and browsers). The participants were asked to assess the adequacy of the pool of nodes and to add any important nodes that were not included. The meaning of each concept was discussed and broad agreement was reached on the nodes to be included.

Participants were asked to indicate the causal relationship between pairs of concepts. Each causal link was assigned a sign (positive or negative) and subjective strength of association from eight alternatives (Positive effects: very strong= 1, strong= 0.75, moderate= 0.5, weak= 0.25 and Negative effects: very strong= -1, strong= -0.75, moderate= -0.5, weak= -0.25).

Each concept variable was be assigned an initial state value and could be held at this value or allowed to vary. The initial state could be set at zero (0), very low (0.2), low (0.4), medium (0.6), high (0.8) and very high (1.0). Most concepts were assigned an initial state value of medium, and were allowed to vary from this state. To test effects of the different management scenarios, management actions were assigned a particular state value, and were "held" at this value. Table 3.1 shows the state values for management actions under the different management scenarios.

Simulation was used to explore outcomes under alternative scenarios, simulated by varying the "held" value of management action concepts, and comparing the output values for the management aims. In this case, the output values of concepts for three different management scenarios were each compared with a scenario where no management was undertaken (base case). The Hot Fuzz software graphed the differences between the concept output values for each management scenario and the base case.

To encourage cross-examination and to limit disagreement arising from language-based ambiguity, workshop participants were invited to discuss each others maps produced in a first round of elicitation. In the light of this discussion, participants were invited to adjust and finalise their maps in a second round of elicitation.

**Table 3.1:** State values for management action concepts under three management scenarios (after Pegler 2009)

| Management scenario | State Values | |
|---|---|---|
| | Burning | Shooting |
| Do nothing (base case) | 0 | 0 |
| Small scale burn & large scale shooting | Medium (0.6) | Very high (1.0) |
| Large scale burn & small scale shooting | Very high (1.0) | Medium (0.6) |
| Large scale burn & large scale shooting | Very high (1.0) | Very high (1.0) |

## 3.3.2. Grassland fuzzy cognitive map

The literature review and experts were used to source a candidate set of nodes for grassland systems. It also would be feasible to run a workshop as above (Pegler 2009), providing a pool of concepts (model nodes) and eliciting extra threats, drivers and/or value, and linking these according to expert knowledge and experience. Figure 3.2 presents the FCM developed for grassland systems. The nodes include potential threats to grasslands (fire frequency, grazing, trampling and soil compaction, cultivation, nutrient and fertilizer inputs, weed invasion, urban development, fragmentation of habitat and edge effects), as well as some of the values that may be impacted (species composition, grassland health – see Section 3.1.1 for a list of indicators).

Fire management is a potential threat to grassland health if it is not managed carefully, taking into account the ecological requirements of all grassland flora and fauna. Grasslands require burning periodically to remove biomass and create inter-tussock spaces for species that rely on these gaps for habitat (Section 2.1). Grazing (Section 2.2) can also be used to create gaps, but will also compact soil and decrease the abundance of palatable species and species intolerant of defoliation (represented by a negative relationship with the 'Species Composition' node). Soil compaction increases with grazing intensity, and in turn impacts on hydrology (reducing infiltration and increasing overland flow and erosion). Other threats are cultivation of the grassland, which may include nutrient and fertilizer inputs, and the introduction of weed species (pasture plants and other species). Urban and industrial development impacts on grasslands by introducing weed species, and increasing the fragmentation of habitat. As grasslands become smaller and more fragmented they are more vulnerable to edge effects (increasing weed invasion, pesticide drift, storm-water runoff, etc).

The links between the nodes (Figure 3.2) indicate the nature of the relationship and the strength of influence.

The model was developed using Hot Fuzz (CSSE, 2008), and can be seen in Figure 3.2. All of the nodes in the model were set at a 'moderate' level (see Section 3.3.1), this formed the 'base case' and specific scenarios were then applied. The first of these scenarios reduced grazing to zero; all other variables were left as moderate (Figure 3.2a). The model simulated the effect of removing grazing and made predictions of the trend in variable states for other nodes, comparing them to the base case. The base case is represented as the black horizontal line in the graphs (Figure 3.2 a, b and c) and predictions made by the model are indicated by the columns in the graph. If the column is above the line this indicates that the effect has been positive, i.e. with grazing removed hydrology is likely to improve, as are grassland condition and species composition. If the columns are below the line it indicates that the effect on this variable is likely to be negative, i.e. gap creation and trampling and compaction have been reduced. The black triangle indicates which of the variables have been manipulated to produce that set of predictions. The columns in the graph are used as a tool for comparison, to contrast the relative effect that management actions will have on the other model variables.

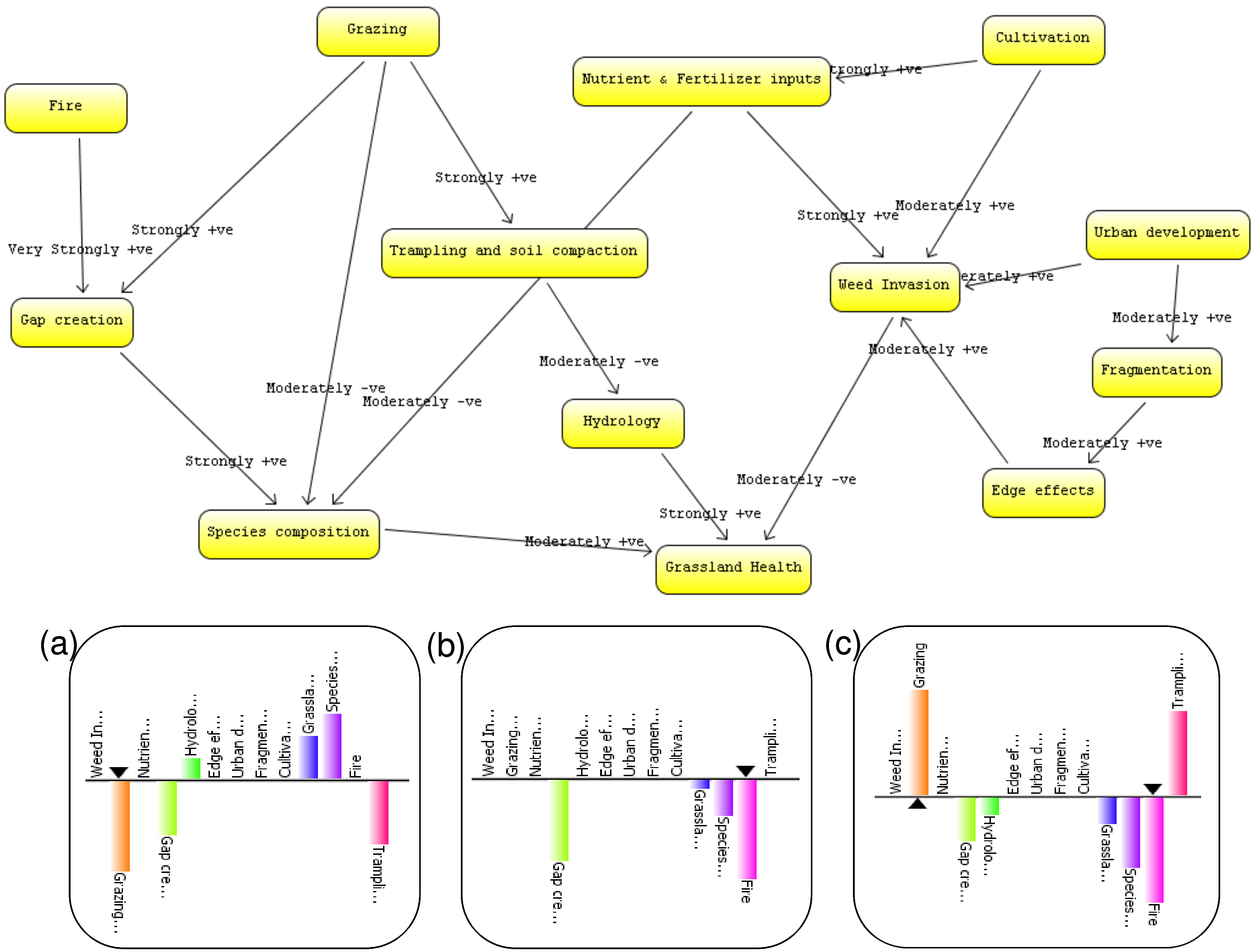


**Figure 3.2:** The FCM for grassland condition. The graphs (a, b, and c) illustrate the predictions simulation) of the model to various management actions, i.e. the removal of grazing (a), discontinuation of burning (b), and a combination of increased grazing and decreased burning (c).

The second scenario reduced the occurrence of fire to zero, and a simulation was run to obtain the model predictions. As can be seen in Figure 3.2(b), this is likely to reduce the level of gap creation in the grassland, which would have a negative effect on species composition, and overall a moderately negative effect on grassland health. The final scenario increases the level of grazing to very high and decreases fire to zero. This simulates a scenario where a grassland reserve is not able to be burnt; this is a plausible scenario where the proximity of factories, freeways, prisons and other facilities excludes burning as an option. The predictions for this scenario are presented in Figure 3.4(c), the likely impacts of this approach are a decrease in gap creation, a reduction in hydrological function, a negative impact on species composition (due to preferential grazing and a reduction in the level of gap creation with no burning), an increase in trampling and soil compaction and a likely overall decrease in grassland health. These are predictions made according to the relationships encoded in the structure of the model, which can be seen in adjacency matrix (Table 3.2).

**Table 3.2:** The adjacency matrix for the model presented in Figure 3.1. This illustrates the relative influence that each of the variables has on each of the other variables.

| | Fire | Graz | Cult | Nut | Tram | Gap | Spp | Hyd | Wee | Urb | Frag | Edge | Gra |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Fire** | 0 | 0 | 0.75 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Grazing** | 0 | 0 | 0 | 0.5 | 1 | 0.75 | -0.5 | 0 | 0.5 | 0 | 0 | 0 | 0 |
| **Cultivation** | 0 | 0 | 0 | 0.75 | 0 | 0 | 0 | 0 | 0.5 | 0 | 0 | 0 | 0 |
| **Nutrient** | 0 | 0 | 0 | 0 | 0 | 0 | -0.5 | 0 | 0.75 | 0 | 0 | 0 | 0 |
| **Trampling** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -0.5 | 0 | 0 | 0 | 0 | 0 |
| **Gap creation** | 0 | 0 | 0 | 0 | 0 | 0 | 0.75 | 0 | 0.75 | 0 | 0 | 0 | 0 |
| **Species composition** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.5 |
| **Hydrology** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.75 |
| **Weed Invasion** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | -0.5 |
| **Urban development** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.5 | 0 | 0.5 | 0 | 0 |
| **Fragmentation** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.5 | 0 |
| **Edge effects** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.5 | 0 | 0 | 0 | 0 |
| **Grassland Health** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

### 3.3.3. Suitability and application

Feedback from experts was mixed. FCMs were considered to be a good method for eliciting cognitive models from different experts. However, other feedback suggested that the FCM did not represent threshold effects or complexity well, and that the limit in the number of nodes would be a drawback.

## 3.4. Bayesian Networks

### Overview

Bayesian Networks (BNs) are a type of graphical probabilistic model, the basis of which is a diagram conceptualizing the ecological system to be managed; this diagram reflects how the system works as an integrated whole (Cain 2001). BNs can be used for many purposes, from illustrating a conceptual understanding of a system to calculating joint probabilities for decision options (Nyberg et al. 2006). The variables in the system, each represented by a node, are linked to parent nodes, on which they are dependant. The arrows between the nodes represent causal dependencies based on understanding of process, statistical or other types of association; they represent the strength of the causal relationship between variables (Pollino et al. 2007). BNs can be used to decompose or partition complex systems into solvable steps, clearly represent values-laden concepts and combine knowledge from different domains and stakeholders (Cain et al. 1999).

In a BN the links between nodes are expressed as dependencies, which are quantified through conditional probability tables (CPTs). These tables consist of a set of probabilities that specify the belief that a node will be in a particular state given the states of the parent nodes (Cain et al. 1999). The links represent cause and effect relationships while allowing for uncertainty, caused by imperfect understanding or incomplete knowledge of the state of a system, environmental variation, or a combination of these factors (Eleye-Datubo et al. 2006). The probabilities in parentless (or input) nodes are assigned according to known frequencies of various states, or based on assumed statistical distributions (Nyberg et al. 2006).

Inferencing (or prediction) using a BN is performed by altering the states of some nodes while observing the effect this has on others (Cain 2001). The node of most interest is the one that represents the management endpoint (grassland condition in this case). The impact of changing the variable is transmitted through the network in accordance with the relationships expressed in the CPTs (Cain 2001). In this way the BN encodes the joint probability distribution over all of the nodes; every time the state of one of the nodes changes the joint distribution is updated through the application of Bayes Theorem (Cain 2001). For each node a histogram represents the probability that it will be in any particular state, calculated from the CPT for this node and probability distributions across the states of the parent nodes (Cain 2001). BNs can be used to test scenarios, by changing a set of variables in a specific manner in order to reflect a set of particular conditions (Marcot et al. 2006). They can be also be used in ‘diagnostic mode’ by setting the endpoint to a particular state and noting the most likely state of all the other variables (Cain 2001, Reichert et al. 2005).

Models can be parameterized with different types of information including:

- Direct empirical evidence (experimental observation of cause and effect);
- Extrapolation (observations made outside of the range at hand);
- Correlation (statistical associations between measures);
- Theory-based inference; and
- Expert judgement.

As other sources of information are often not available the literature and expert judgement are frequently used (Burgman 2005). Expert judgement is not a substitute for definitive scientific research; however it can provide useful insights for policy and decision makers while research to produce more definitive results is ongoing (Morgan et al. 2001). A carefully devised and calibrated probability network is ideally suited to communicate the interface between scientists, stakeholders and decision-makers when data is basic or absent and uncertainty is considerable (Reckhow 1999a).

The BN may be less suitable for the early stages of elicitation, the causal map (Section 3.2) would probably be more appropriate, and easier to use for the basis of initial discussions. In a staged approach to model building would use BN later in process, to capture the complexity and subtleties of system.

### Strengths

BNs have distinct advantages for the modelling of ecological systems, and to aid in decision support. They are able to incorporate qualitative and quantitative data, and judgements elicited from scientific experts and other stakeholders. They are able to explicitly connect multiple system processes and be used to identify key drivers when system issues may be complex and dynamics may be poorly understood (Pollino et al. 2007). BNs are good at representing thresholds, which are used to form the cut-off points represented in the state names; break-points may be drawn from the literature, experimental results or expert opinion and represent biologically relevant thresholds. Node states are expressed in measurable terms suitable for testing and inclusion of empirical data once available (McNay et al. 2006). BNs are able to incorporate management actions as well as threats. They also fit into the cycle of adaptive management, as they are iterative and adaptive; the model can be updated when new information becomes available via Bayesian inference (Nyberg et al. 2006).

BNs are able to operate at a number of different scales within the same network (Borsuk et al. 2003). They are decomposable, allowing the conditional probabilities to be estimated using separate sub-models (Reckhow 1999a). BNs can be used to identify the variables that have the greatest influence, but are understood the least, guiding future data collection.

BNs represent uncertainty in the way in which the system functions, using conditional probabilities rather than deterministic relationships (Stow et al. 2003). Biological processes are complex and stochastic, making representation by probability distributions appropriate; the subsequent probabilistic predictions give a more realistic impression of the chance of achieving the desired outcome (Borsuk et al. 2004).

Clear inference allows scenario testing to examine the interaction of different management actions (also interactions with variables that are not able to be managed, such as climate change). Feedback is immediate and this helps with elicitation; scenarios can be run quickly so that the implications of management interventions are rapidly understood (Reckhow 1999a). BNs allow clear articulation of threats, hence decisions to be made by management team can be focused, the analytical rationale for management options defendable, and the protocol for monitoring success and failures explicitly established.

Probabilistic models (such as Bayesian Networks) allow the description of key mechanisms without the full information requirements of process-based models (Borsuk et al. 2004). These models can assist with multiple stressor problems, are able to incorporate information with high uncertainty, including poor or incomplete understanding of the system, and can include empirical data and expert opinion (Hart et al. 2006). Finally, and perhaps most importantly, these models provide predictive link between management actions and ecosystem response.

### Weaknesses

The disadvantages of BN models are that they cannot represent feedbacks or time-dynamic functions (Nyberg et al. 2006), and if quantitative data is not available, the relationships between variables must be elicited using expert opinion (Barton et al. 2008). Building CPTs for a large BN creates a large elicitation burden (compared to FCMs where the relative strength of the link between variables is elicited). The advantages of BN model development are offset by the cost of obtaining reliable probabilistic data (Barton et al. 2008).

### 3.4.1. Grassland Bayesian Network

The causal map (Section 3.2) was used as a basis for the grassland BN. As for the FCM (Section 3.3), the information could also have been derived by eliciting the model structure (threats, processes and values) in a workshop setting, which facilitates the sharing of alternative perspectives and encourages collective understanding.

Figure 3.3 presents the BN developed for grassland systems. The nodes include potential threats to grasslands (fire frequency, climate change and drought, grazing, trampling and soil compaction, cultivation, nutrient and fertilizer inputs, weed invasion, urban development, fragmentation of habitat and edge effects), the values that may be impacted (species composition, grassland health – see Section 3.1.1 for a list of indicators) and some of the potential management interventions (e.g. native species re-introductions, scalping of the top layer of soil, carbon addition). Due to time constraints expert opinion and the literature has been used to parameterize the BN for grassland condition. The modelled results are consistent with field observation; however, the actual probabilities in the BN should be regarded purely as estimates and used with caution (Batchelor and Cain 1999).

Planning for fire in grasslands systems is particularly important, as indicated by the links between the fire nodes and other nodes in the network. Among the things to be considered are the amount and condition of fuel (related to the season of the burn), the likely response of weed species and native species (Section 2.1), fauna (e.g. *Delma impar*) and threatened plant species. The states in the fire node represent critical thresholds for *Themeda*-dominated grasslands, which should be burnt at least every 11 years to remove *Themeda* biomass (Lunt and Morgan 1999). The formation of gaps, by creating opportunity for inter-tussock species to germinate, increases species diversity in the grassland, and contributes to the heterogeneous structure that is considered to be a sign of grassland in good condition (Section 3.1.1).

Grazing may also be used to create gaps, as reflected in the BN, and may be necessary when grasslands are close to infrastructure which makes the use of fire difficult, or if seed production of key species is low due to drought. Grazing animals will have an impact on soils, increasing compaction and decreasing infiltration. Grazers will also tend to introduce weeds to an area and will contribute nutrients to the site.

Urban development contributes to habitat fragmentation, which can make the grassland more vulnerable to pesticide drift, weed invasions and storm-water effects. Carbon addition improves the carbon-nitrogen ratio but feedback suggests that it would only be a small-scale management action, e.g. used when introducing a threatened plant species to make conditions less favourable to weed species (conferring a competitive advantage to the native species). The purple nodes in the BN (Figure 3.3) represent to the fauna which reply on grasslands for habitat, and how threats and potential management actions may impact on them.

The BN allows all of these factors to be considered together as part of the same modelling system, which helps to weigh up the advantages and disadvantages of a particular management intervention (or a combination of interventions). The BN could also be used to incorporate value judgements by using utility nodes (nodes which incorporate costs and benefits, as judged by experts) based on the preferences of key stakeholders.

### 3.4.2. Suitability and application

There was considerable interest in the BN for grasslands. One of the most useful features of this modelling method was the ability to test scenarios and obtain immediate feedback on the likely effect of specific management interventions, or multiple interventions. The main disadvantage was the amount of time and effort required to develop the CPTs, though in this model the model was parameterised using the literature, and evaluated using expert feedback (which was largely positive). Another potential disadvantage is that to interact with the model requires specialist software (Netica, www.Norsys.com). This problem could be overcome by developing a web-based version of the BN, where node states can be changed to test the impact of different scenarios on the nodes of interest.

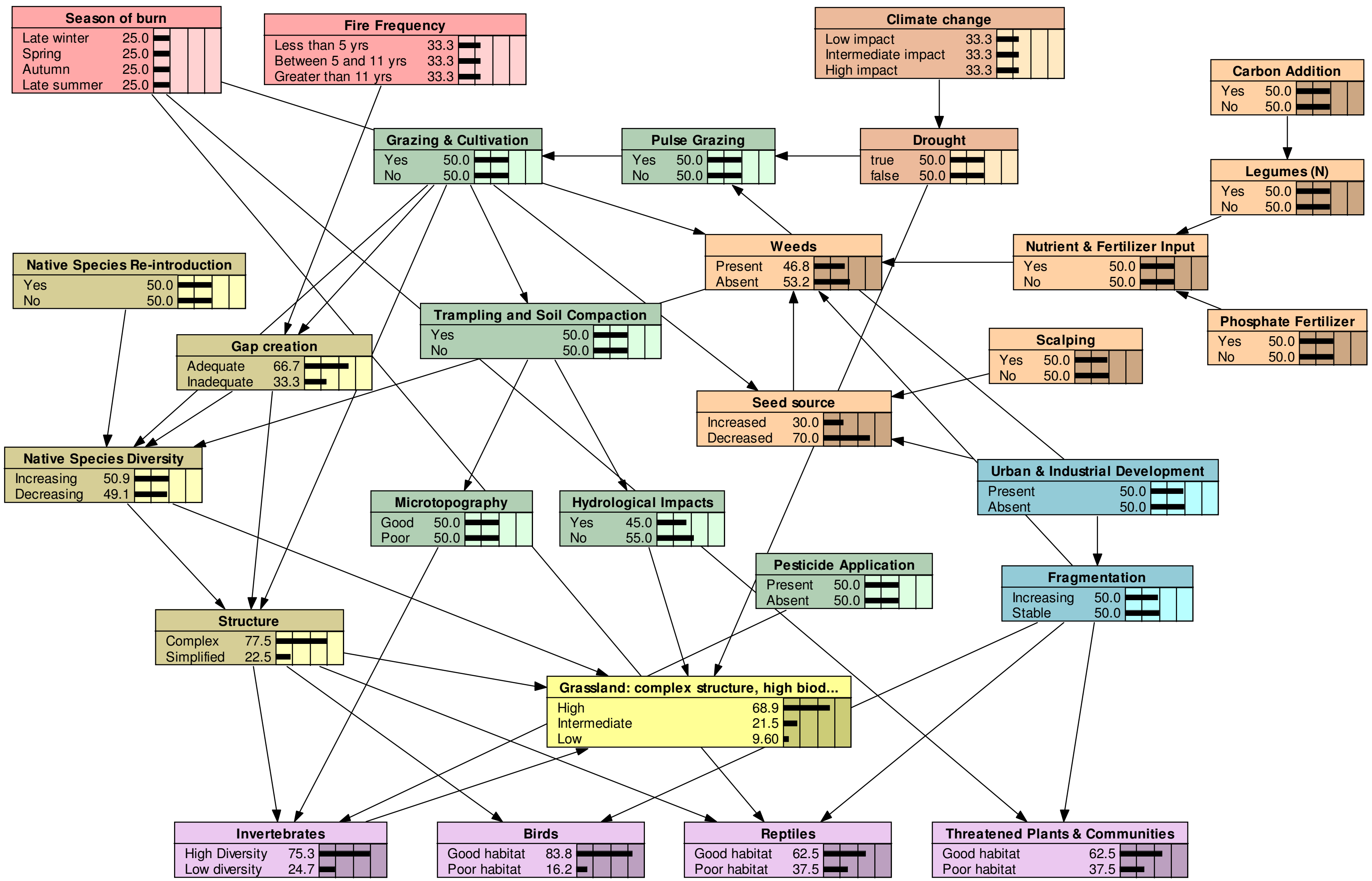


**Figure 3.3:** The BN for grassland condition, which includes the threats to grasslands (and the fauna that depend on grasslands for habitat), as well as some of the management interventions.

## 3.5. State-Transition Models

### Overview

State-transition (ST) models are simple diagrams that represent observed or theoretical states; arrows represent the observed or theoretical transactions between these states (Jackson et al. 2002). ST models are used to conceptualize complex behaviour of dynamic systems, and are useful in situations where system behaviour is non-linear, and classical (linear) climax theory is not well-suited (where an ecosystem is not able to be described by a single equilibrium community and a deterministic pathway for succession) (Westoby et al. 1989).

ST models are qualitative; they have the capacity and flexibility to accommodate various types of knowledge and information associated with vegetation management. The primary components of ST models are states, transitions and thresholds (Stringham et al. 2003) that are determined by the resilience of the ecosystem and its response to primary ecological processes. They are intended to function on the basis of managerial, rather than ecological criteria (Westoby et al. 1989). Information required to build these models include the following (Ibid.):

- The potential alternative vegetation states on a site;
- The potential transitions between states on a site;
- The opportunities to achieve favourable transitions between vegetation states; and
- The opportunities to avoid unfavourable transitions (hazards).

Vegetation states are identified based on species composition, structure and abundance, they remain relatively stable for long periods of time and only change because of natural or human disturbance. States are not truly stable, but are constant over time scales relevant to management regimes (Wilkinson et al. 2005). Within a state there is room for variability in species composition that is a natural part of community dynamics. Components used to describe states can include abiotic and biotic elements such as soil base and vegetation structure (Stringham et al. 2003).

Model formulation includes determining which of the states are linked, describing the transitions, and usually also delineating one or more desirable states from those that are less desirable (Briske et al. 2003). Direct transitions or linkages do not occur between all states. Transitions that move away from the desired state toward more highly degraded states have been identified as obstacles to restoration, whereas transitions that move from degraded to less degraded states are identified as restoration opportunities (Wilkinson et al. 2002).

Indicators of the state of an ecosystem may include perennial plant cover, shrub density, amount of bare ground, soil compaction, surface soil stability and/or plant composition data.

Ecological indicators can be used to anticipate transitions, but they must be properly defined, and include reference values for quantitative indicators, description of changes that suggest approaching transition, and rigorous documentation of theory and assumptions underlying them (Bestelmeyer et al. 2003). Indicators are used to connect field observations with theoretical expectations and management responses. In many cases transitions occur one patch at a time, occurring first in areas most sensitive to change due to slight variations in soils or landscape position (Ibid.).

A workshop composed of people with expertise in a plant community is a useful means for the identification of vegetation states. Feedback from land managers and end-users can help validate and complete models designed by workshop participants. ST models can guide decision-making by identifying possible outcomes of each current state and the conditions required for transition between states. Undesirable states can be identified, as well as the activities that facilitate transition to these states, and a proactive approach taken to prevent their occurrence (Westoby et al. 1989).

A part of this modeling system is the notion of threshold changes which bring about the transition from one state to another. Smaller changes that do not cross these thresholds revert to pre-disturbance conditions without management intervention. Thresholds mark the point where a system will not naturally return to its previous state, and management intervention is required (Wilkinson et al. 2002), which may include seeding, shrub control, recovery of soil stability or hydrological function (Bestelmeyer et al. 2003). When a threshold has been crossed, the sites primary ecological processes are modified, resulting in a different potential set of plant communities. Primary processes include hydrology (capture, storage, and redistribution of precipitation), energy capture (conversion of sunlight into plant and animal matter) and nutrient cycling (cycling through the physical and biotic components of the environment) (Pellant et al. 2005).

The intent is usually to determine the main processes that cause transitions toward or away from desired restoration states so management decisions can be made (Wilkinson et al. 2005). A given state will persist until an event or processes cause changes in the types of species and the system is forced to reorganize (Holling 1973, Walker 1995). Restoration efforts aim to place a degraded community on a trajectory toward a desired state. Transitions occur when one or more constraints are altered by external factors, and this change catalyzes changes in feedbacks and produces shifts in vegetation structure and soil properties. It is possible to ‘push’ an ecosystem into a particular direction, e.g. introduce propagules, reduce the cover of undesirable species, manipulate fire frequency and/or use herbivores that target specific species. Intensive management interventions (e.g. gully stabilization) may also be necessary.

**Strengths**

ST models can accommodate multiple equilibria (alternative stable states) or non-equilibrium (no stable state) dynamics (Jackson et al. 2002). They provide a practical way to organize information, and can accommodate non-linear behaviour. Laycock (1991) suggested that the use of ST models increases the feasibility of management programs and reduce false expectations by more realistically modeling biological systems. ST models may be used to develop testable hypotheses in which transition probabilities could be theorized or empirically generated (Jackson et al. 2002). ST models incorporate current knowledge, are adaptive, can provide guidance for restoration research and conservation management in degraded systems (Wilkinson et al. 2002).

**Weaknesses**

Problems can arise in the use of ST models when errors occur in the identification of states and their classification, and in the occurrence of unusual transitions (Allen-Diaz and Bartolome 1998). These types of models may require detailed long-term data, which is usually not available (Allen-Diaz and Bartolome 1998).

## 3.5.1. Grassland State-Transition Model

Literature review and expert feedback were used to source a set of states and transitions for grassland systems. As for the FCM (Section 3.3), a feasible alternative would be to run a workshop to elicit the structure of the model (states, transitions, thresholds and potential management actions).

Grassland changes in response to grazing have often been found to be discontinuous, irreversible or inconsistent (e.g. Lunt et al. 2007). Alternative stable states, discontinuous and irreversible transitions, non-equilibrium communities and stochastic effects are able to describe these systems more realistically (Westoby et al. 1989). Transitions between states may be triggered by weather, fire, a change in stocking rate, introduction or removal of weed species, and/or fertilizer use. Grassland ST models require a list of possible alternative stable states, transitions (weather events, management), opportunities (climatic circumstances under which actions such as fire, heavy grazing, removal of grazing etc, could produce a desirable transition) and hazards (climatic circumstances under which failure to burn, heavy grazing, etc could produce an undesirable transition) (Westoby et al. 1989).

Figure 3.4 presents the ST model for the management of fire in a *Themeda*-dominated grassland; the blue boxes represent the different states, the green ovals the possible management actions and the yellow diamonds the moderating effect of the season of the burn. The states in the model are a theorised pre-settlement historical state, a degraded state that has resulted from heavy grazing, soil disturbance and nutrient inputs (low floral biodiversity, degraded soil structure and dominated by weed species), a degraded state that has resulted from a lack of disturbance (high *Themeda* biomass, low floral biodiversity), a state that has been burnt (but differs in its features depending on the season of the burn) and a healthy grassland which has resulted from regular burning and the re-introduction of species that have disappeared due to non-persistent seed bank (the 'desired' state). The healthy state requires periodic burning (inter-fire period less than 11 years) or will revert to a degraded state.

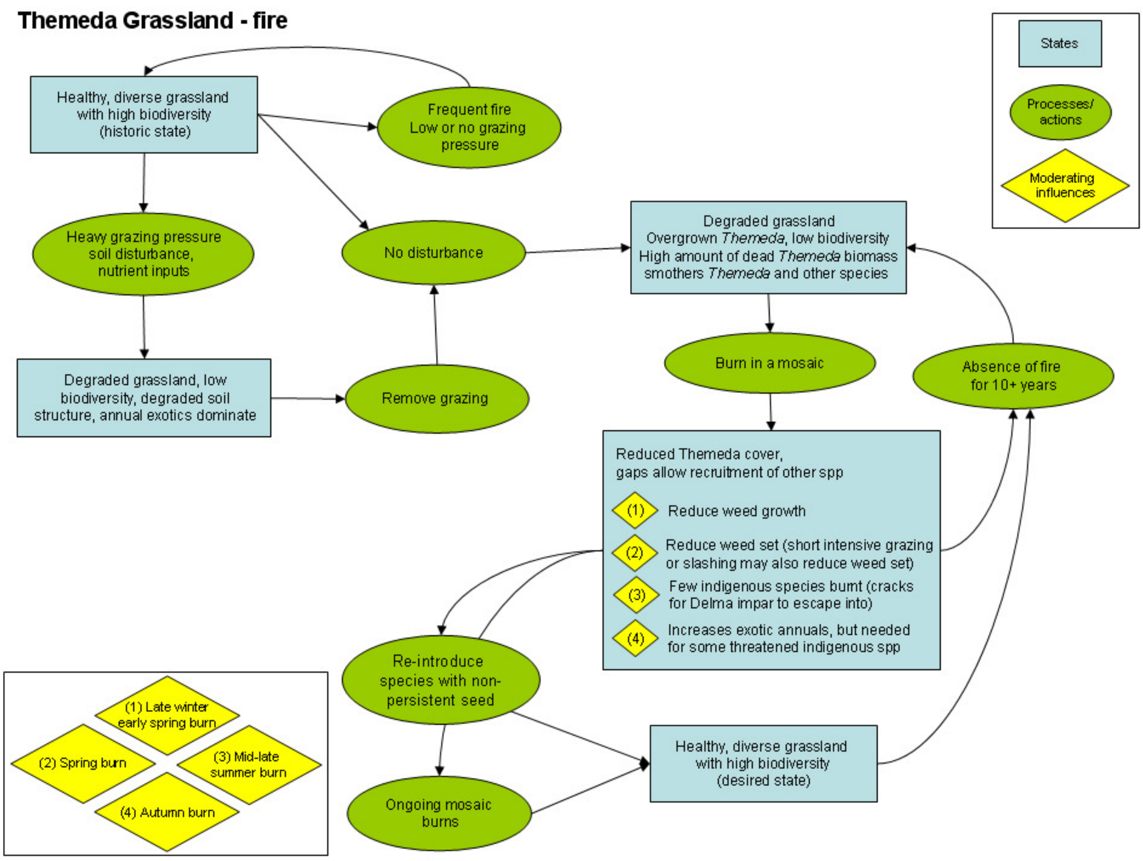


**Figure 3-4:** State-transition model for different fire regimes in *Themeda*-dominated grasslands. This model outlines, in general terms, the different states of a *Themeda*-dominated grassland, and the likely consequences of various management actions related to the fire regime.

Figure 3.5 presents the ST model for the management of grazing in a *Themeda*-dominated grassland. The states in the model are the theorised historic state, a degraded state that has resulted from heavy grazing, soil disturbance and nutrient inputs (low native biodiversity, degraded soil structure and dominated by weed species), and a less degraded state that has resulted from a reduction in grazing pressure (improved biodiversity and vegetation structure). This model incorporates the use of grazing to manage biomass and pulse grazing for weed control. This may be used in situations where burning may be too difficult or dangerous (e.g. proximity to freeway).

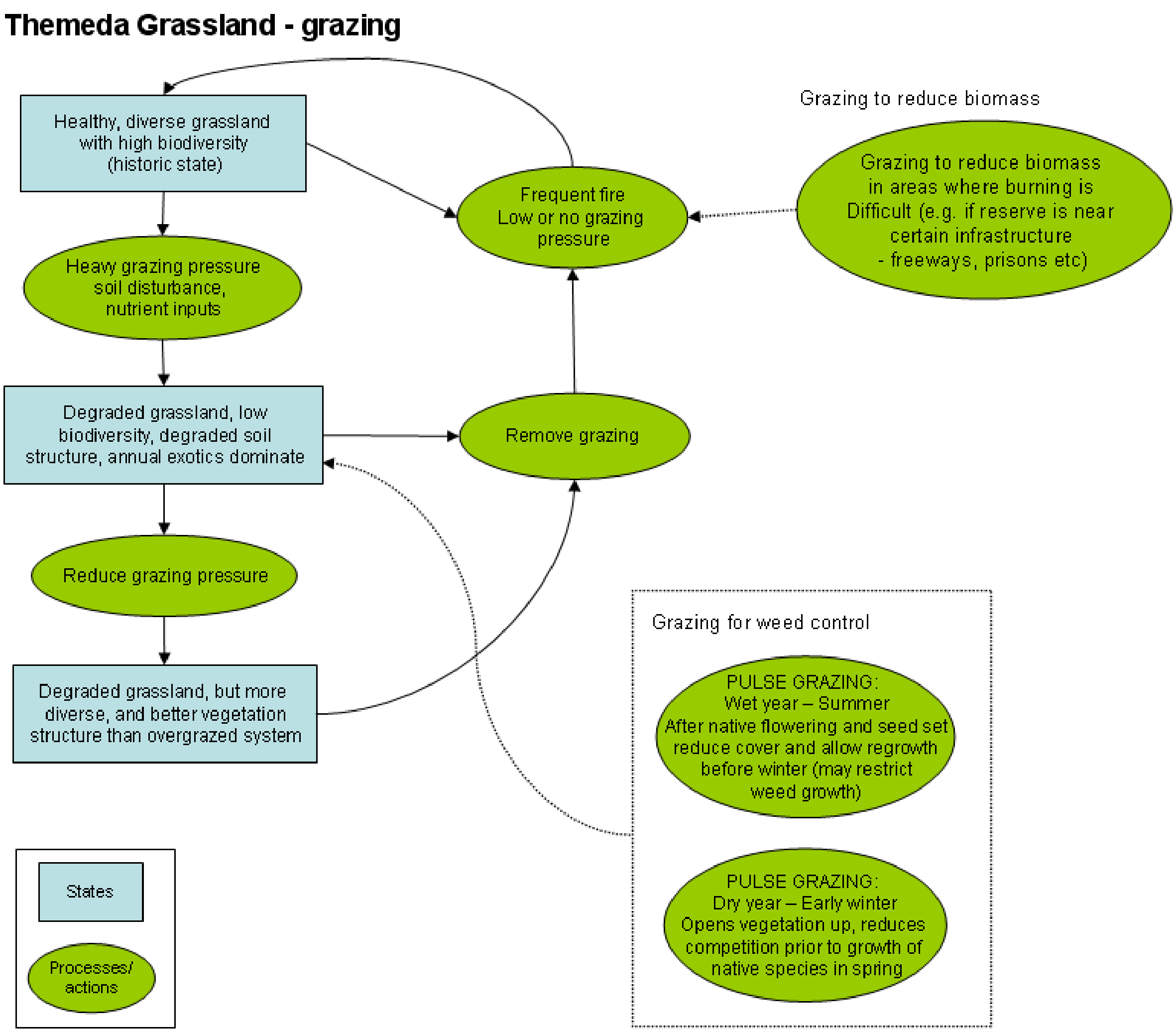


**Figure 3-5:** State-transition model for different approaches to the management of grazing in Themeda-dominated grasslands. This model outlines, in general terms, the different states of a Themeda-dominated grassland, and the likely consequences of various grazing strategies.

Figure 3.6 presents the complete ST for management of a *Themeda*-dominated grassland, incorporating the effects of grazing and fire. It should be noted that the ST models presented in this project are comparatively simple, and do not go into as much detail as other grassland ST models which typically include details of cover and abundance of important species and/or structural vegetation types (e.g. Wong and Morgan (2007) use the percentage cover of *Themeda*, native annuals and perennial shrubs in their example of a ST model for grasslands).

Figure 3.7 presents a ST model for a non-*Themeda* dominated grassland. These systems are characterised as having lower resource availability and a lower level of productivity (Prober and Thiele 2005). As a result these systems are more vulnerable to the impacts of grazing, which damage soil structure and result in increased erosion, decreased infiltration, more extreme microclimates and the destruction of soil crusts, which provide critical ecosystem services. These grasslands are dominated by wallaby, spear and windmill grasses, and probably a denser shrub component (Prober and Thiele 2005).

Grazing in these systems causes a loss of palatable species and a simplified vegetation age and size structure. Fire is an important source of disturbance in these systems, but is not required at the same frequency as in *Themeda*-dominated grasslands (Eddy 2002). Gap formation in these systems may also be created by the occurrence of drought (Westoby et al. 1989), and may be maintained by the activity of faunal species.

### 3.5.2. Suitability and application

Feedback from experts was generally positive. These models would be well-suited to provide the next level of detail in the causal map (Section 3.1). These models could useful to pass knowledge on, makes it clear what the likely outcomes are from specific management actions. Other feedback suggested that these models could be misleading without extra knowledge and context. This expert stressed that ST models are a snap-shot and that a person would need to see a grassland system over a long period of time to get an understanding of its different states.

## Themeda Grassland

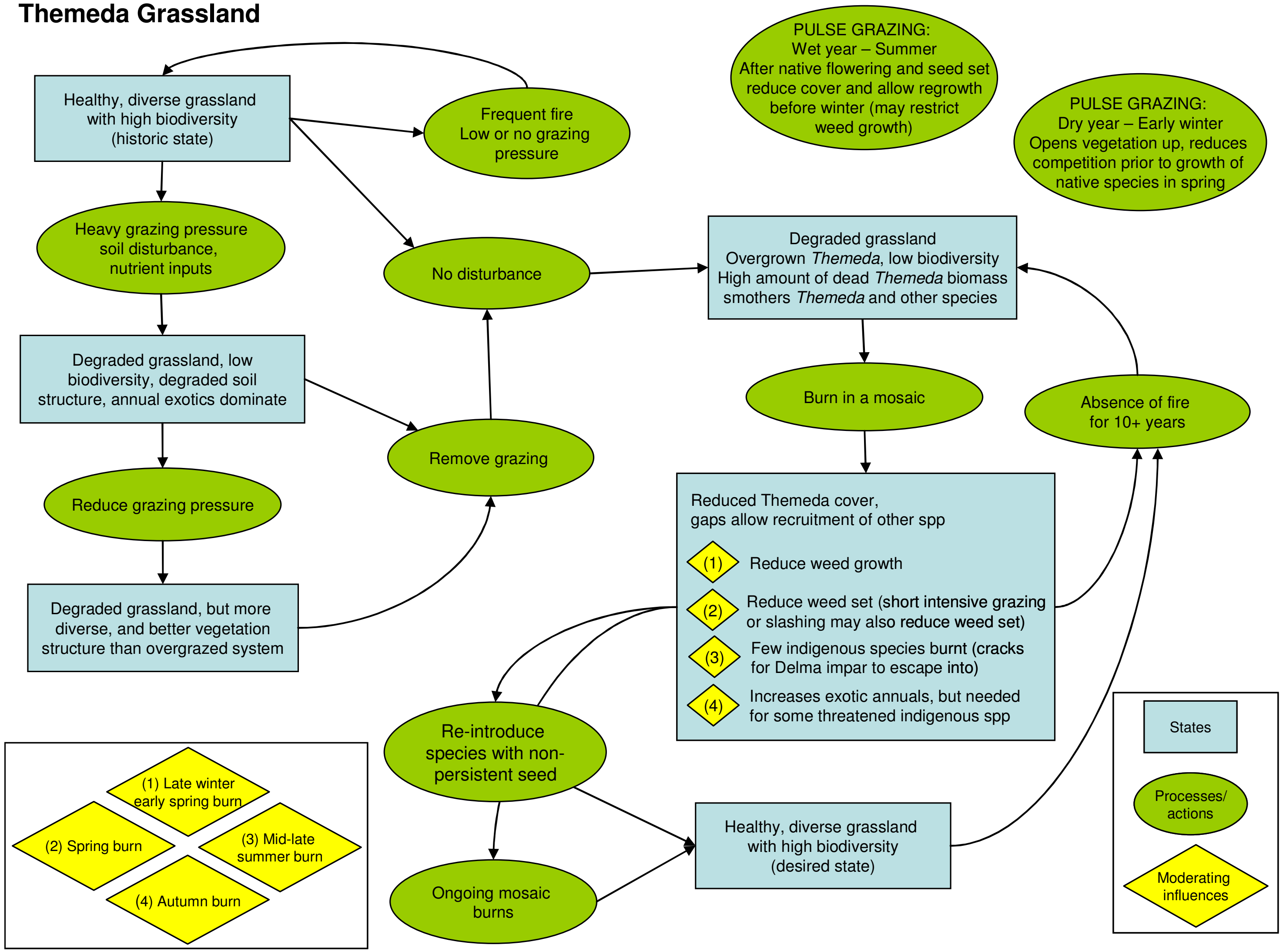


**Figure 3-6:** The State-Transition model for a *Themeda*-dominated grassland (including fire and grazing management), illustrating the possible states of the grasslands, potential management actions, and their likely effect.

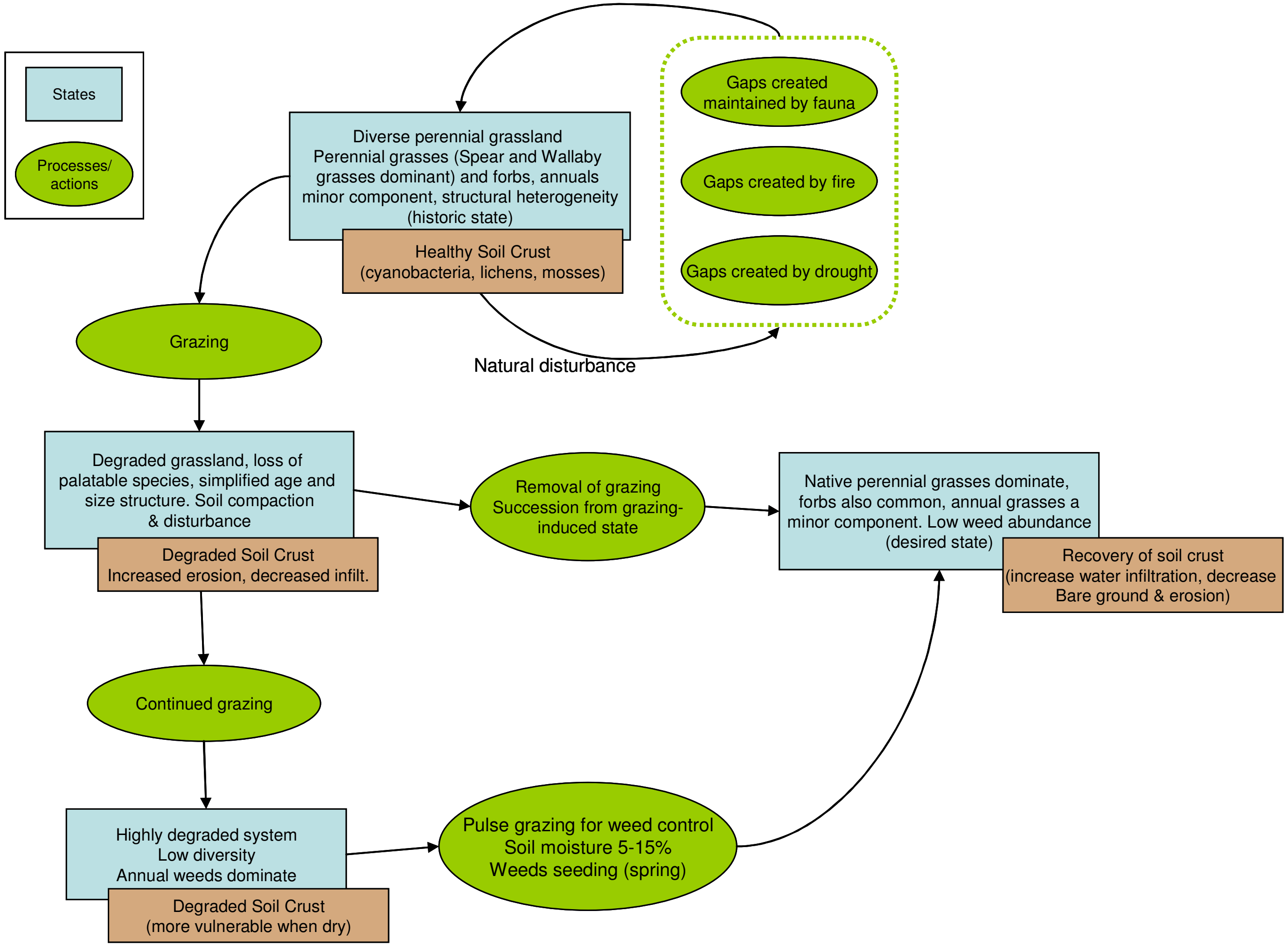


**Figure 3-7:** The State-Transition model for non-*Themeda* grassland, illustrating the possible states of the grasslands, potential management actions, and their likely effect.

## 3.6. Feedback regarding applicability of models

The various models developed for grasslands were presented to a number of experts to elicit feedback. A model walk-through was conducted with experts individually. Overall feedback was very positive.

**Mark Antos, Environmental Scientist, Fauna Department**
**Research management and effectiveness, Parks Victoria**

Mark was approached in order to obtain feedback on which of the elements (of which of the models) could be usefully applied to the Signs of Healthy Parks (SHP) Program, which was one of the main aims of this project. The goals of the SHP program are as follows:

- Assess the trends in the condition of natural values and threats in parks through monitoring appropriate quantitative indicators;
- Systematically evaluate the effectiveness of management actions;
- Provide early warning of emerging threats; and
- Provide data to assist future management and decision-making.

What follows is a summary of Marks's feedback. One of the biggest advantages in the development of ecosystem based conceptual models would be to support decisions relating to prioritization of threats, values and ultimately management actions. Currently the assessment of the condition of natural values, and the effectiveness of management actions, occurs on an individual, park by park basis, as follows. All current literature on the park is assembled, this will include any work that has been carried out in the past, such as management plans, conservation action plans, risk assessments and a list of values, threats and things to monitor is drawn up. The next step is to consult local staff, and elicit from them the values within the park, the threats to values, what should be monitored to detect trends in the condition of these values and the effectiveness of management (would usually get around 10-20% of extra information from this activity). Typically this process results in the creation of a comprehensive but very long list of values, threats and potential monitoring targets.

Each park will typically have a number of different ecosystems; the ecosystem conceptual models (when fully developed) would help to quickly identify the likely threats, values, processes and important ecosystem drivers. Ecosystem models would also help to identify the things on the list of potential threats, values and monitoring targets that should be followed up, making the full list more manageable. From this we could identify the most important indicators, look at the park management plan, and align efforts with resources that are available. This approach would help to decide on the amount of effort and resources that should be assigned to different threats and values, and what should be monitored to gauge management effectiveness [*A White: in line with PV's Environmental Management Framework (EMF) which aims to mange natural values in a systematic way, taking into account limited knowledge of systems using a risk management approach (State of the Parks: Effectiveness of Natural Values Management, 2007 page 5*)].

Conceptual models will be instructive from the point of view of identifying the most important parts of any process by noting which of the elements in the model are most heavily linked, these being the most important processes, threats and management alternatives. This should help to determine if the right indicators have been chosen; i.e. is the program targeting the most important values/threats/drivers as identified from the conceptual models (which reflect all of our understanding and knowledge of the system)?

Mark also highlighted that ecosystem conceptual models would be very useful in communicating the strategy behind monitoring and management programs to on-ground staff, whose involvement is crucial in carrying out management actions and monitoring. Specifically, ecosystem models would help to communicate why we would monitor specific values and not others, and why we would manage specific threats and not others, by making transparent the method and approach of the prioritization process. They would also demonstrate how different ecosystem components interact, and how different conservation and management programs overlap and interact.

The areas in which the ecosystem conceptual models may be applied to planning for the SHP program are outlined in Figure 3.8.

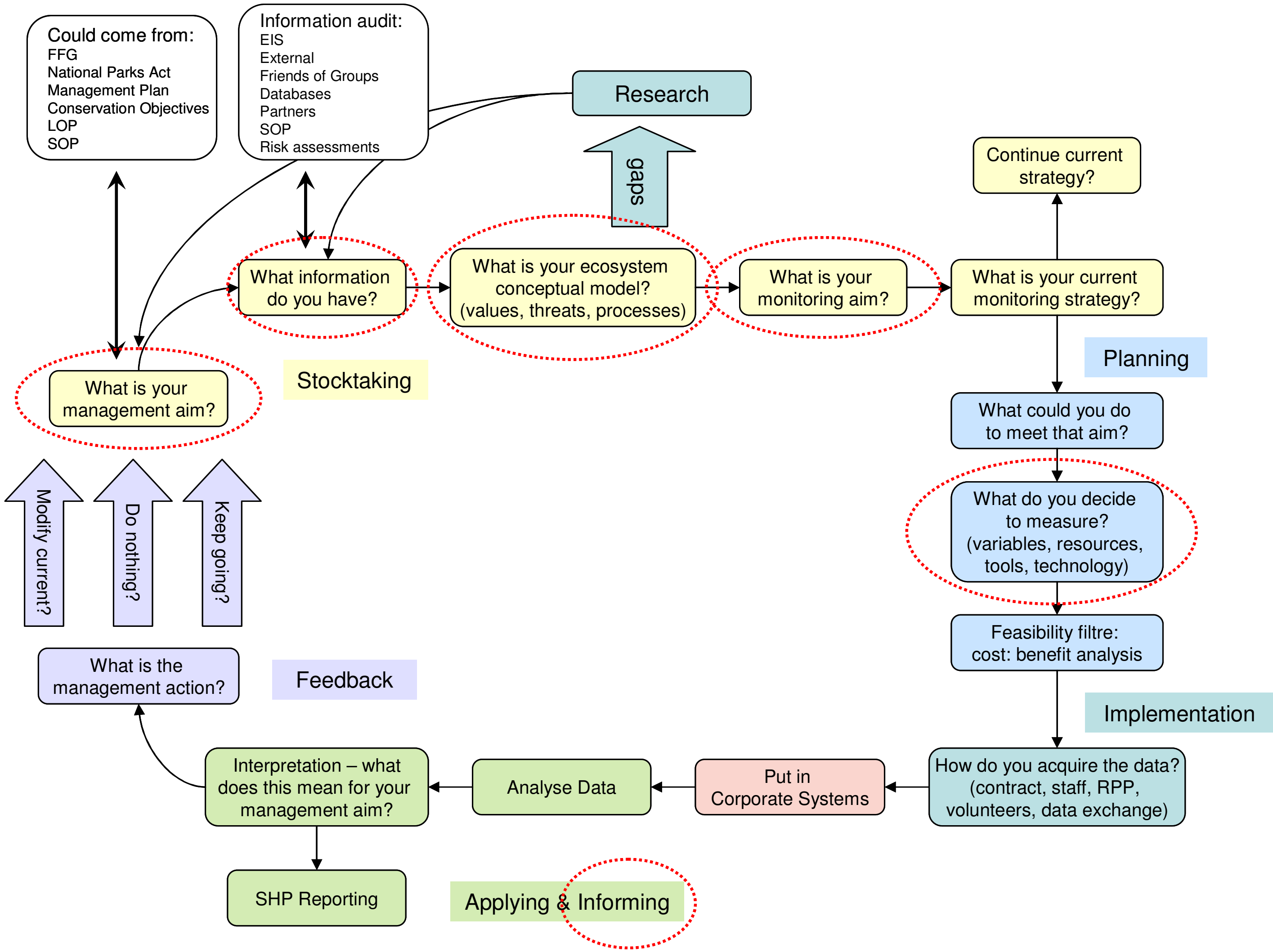


**Figure 3-8:** Signs of Healthy Parks (SHP) framework (Mark Antos, PV 2008). The red ovals indicate where the ecosystem conceptual models may provide input into the SHP framework.

**Fiona Smith, Program Manager Biodiversity and Pests**
**Healthy Parks Program, Parks Victoria**

Fiona has over 10 years experience in managing grasslands, being responsible over that time for 13 of the reserves in the west of Melbourne. Fiona was approached in order to obtain feedback on which of the models could be usefully applied to grasslands at the park level, for on-ground management. The following is a summary of Fiona's feedback.

Grasslands are very responsive and require active management, what you see at a site is the result of the last 10-20 years of management. Therefore it is particularly important to correctly predict the consequences of management actions. Conceptual models could be used for decision support, in prioritizing different species (including threatened species), and choosing one intervention over another. The models would be useful as a visualization tool, would help to get a fuller picture of the system, and help with thinking through influences and likely consequences and outcomes.

Causal models would be particularly useful for communication, for example when interacting with:

- Councils and other external agencies when determining the offsets for urban development projects (reserve placement and design). Particularly in communicating an understanding of the impacts of development on grasslands. The example that Fiona used was that burning of grasslands for management (biomass removal) may not be possible if prisons or freeways are built nearby.
- New staff, and also staff that might have a more operational background (rather than ecological), particularly in passing on knowledge when staff leave a particular park (reducing the impact of staff turnover). Conceptual models would be useful in highlighting the complexities that must be considered when managing a grassland system, including the management options and different states that are possible.
- Useful for making funding cases internally (justifying recurrent expenditure, etc).

Would like to be able to highlight different sections of the model in order to demonstrate specific impacts or consequences for carrying out (or not carrying out) certain management actions (light up a path to illustrate a process/impact). In order to have trust in the model it should have PV input throughout the model building process.

**Brendan Sullivan, Ranger in Charge**
**Organ Pipes National Park, Parks Victoria**

Brendan is the Ranger in Charge of the Organ Pipes National Park, and was approached to obtain feedback on which of the models could be usefully applied to on-ground management. The following is a summary of Brendan's feedback.

A positive feature of the causal model is that it can include all factors and therefore create a complete picture for consideration when making management decisions. Climate change and the effect that this is having on drought is an important impact and should be included (this was added after discussions with Brendan).

The BN, in that it is interactive, can test different scenarios and determine the likely response of key system features. This would be very useful when considering the consequences of alternative management actions. The BN would be particularly useful for making sure all of the impacts of fire are considered before a burn is undertaken, including the effects on target species (biomass removal, weed control) and the impact on threatened flora and fauna. Some of the species that need to be considered when in planning burns (particularly the timing) include the following:

- Golden Sun Moth – emerge in October and November, this is when they take flight and breed.
- *Delma impar* – late summer is the only time when there are sufficient cracks in the ground for refugia.
- Spiny Rice flower – not enough known about the fire ecology of this species, needs more investigation into the sensitivity of different life stages and responses to fire (knowledge gap).
- Sunshine Diuris – may weigh up the pros and cons of a burn and decide to sacrifice a seasons seed for this species as there is evidence that populations can recover well from this.

A warning could come up for impacts on particular species if a burn was planned at a particular time of the year (e.g. if *Delma impar* is present a warning would come up for burns except if they were in late summer).

Including the current condition of the grassland would be really useful, so that the predicted system response for management actions (or combinations of actions) could be examined and compared to current condition. Ballarat University produced some vegetation condition indices and a framework for measuring condition; this could be incorporated into the model. A list of 'desirable' attributes in the grassland, i.e. the things we are managing for, would make the model more practical.

Grasslands often need an alternative to fire. Grassland species are not currently responding well to fire; this is probably due to a succession of years where it has not been wet enough for the species to produce a good crop of seeds and the seed bank has been depleted. Pulse grazing could be used to reduce biomass instead of fire (leave standing stock, rejuvenates tussocks rather than relying on species to come back from seed). However, this approach is not effective to manage some pest species, such as Serrated tussock. It can also be very difficult to carry out burns because of nearby industrial facilities, especially petrochemical companies.

Herbicide management is very important, rely on this to treat weeds after fire (e.g. Serrated Tussock and Chilean Needle grass), chemical is water activated, amounts applied have to be carefully considered so as not to effect non-target species.

Storm-water impacts on some grasslands are considerable, this makes management very difficult. Some areas may need to be excised from reserves as they have become too degraded. Impacts include erosion (water coming off escarpment undercuts the edge, erosion hard to control), and transport of weeds into the site. There may be historical uses to consider, e.g. model plane club, has a runway area that is mown very short and has had all rocks removed, can not rotate use as they would want to remove rocks from new site (not acceptable from the point of view of fauna).

There are some practical considerations when planning for an effective fire regime. The size of the burn and the season of the burn are important. Sites must be accessed with equipment for preparation and management of a burn, because of this smaller mosaic burns were difficult to implement. The approach now is to burn in larger patches, and this seems to work. Pulse grazing is a good alternative in locations/circumstances where the grassland can not be burnt

Brendan expressed the need for a decision tool to help predict the likely response of key grassland species to different management regimes. Brendan highlighted that even with what we know now about ecological and individual species needs mistakes are still made in how grasslands are managed. He suggested a management tools that would highlight the impact for e.g. burning at a particular time of year (a warning may come up that this will be detrimental to specific species, or the system as a whole).

### Feedback from Healthy Parks Branch and Research and Management Effectiveness Branch

The following is a summary of feedback that took place at a meeting where the pilot conceptual models for grasslands were presented.

The aims of the conceptual model project are to:

- Reflect basic understanding of how each ecosystem works.
- Identify and rank the important drivers (threats, processes) of system condition.
- Promote understanding and consensus and also to identify areas of disagreement/division.
- Support decision making – to help make sensible decisions about how to manage a system.
- Facilitate communication internally and externally, both to pass on information about the system, also to communicate the rationale behind management actions.
- Identify the elements of the system to monitor to evaluate management.
- Identify important knowledge gaps and areas of uncertainty to guide research.

As our ultimate aim is to hold on to high value patches of habitat and threatened species. Modeling the main ecosystem types may help to understand better what issues/drivers/management interventions might be and promote integration of management programs. For example, models might help to determine what to manage, when evaluated along with all other threats and issues, may decide that a specific management action is unnecessary, maybe even counter-productive. This is especially important as resources will always be limited, and prioritization necessary. The ecosystem models would be useful to as input into the development of park management plans. The Levels of Protection currently has fairly loose objectives; these could be refined by development of conceptual models, which would help to work out values, and what processes need to be managed.

The ecosystem models would be conceptual rather than spatially explicit, and therefore able to be applied at a number of scales. This will be an advantage if objective setting could occur at state-wide scale, and the process could then be scaled down to the park level, as part of management planning.

## 3.7. Time lines for modeling stages

The process of developing conceptual models for each of the ecosystem types would begin with research and review of the literature. Each of the ecosystems will have its own list of values, these may be extracted from park management and conservation plans, the literature or they may be developed in a workshop setting (see Carey et al. 2007). The most productive approach would probably be a combination of all of these approaches. Once the literature has been reviewed and the values and threats identified the first draft of the conceptual model would be developed with one or more of the modelling methods.

**1.** Research of literature: 2-4 weeks

**2.** Literature review: 2 weeks

**3.** Workshop to elicit values (includes identifying appropriate model/management endpoints, and produce a comprehensive list of threats) 2 days

**4.** First draft of conceptual model (may be rudimentary or quite well progressed) 1-2 weeks

**5.** Causal map development

a. Pre-consultation preparation (develop a map to use in workshop, and some sub-maps, to demonstrate the ideas behind the process and software) 1-2 days

b. Expert workshop / consultations (get experts together, talk about methods and principles of causal mapping, practice on software in preparation for further work after workshop) 2 days

c. Ongoing management of model refinement process 2 days

d. Model aggregation, expert feedback 1 week

e. Write-up 1-2 weeks

**6.** Fuzzy Cognitive Map development

a. Pre-consultation preparation (develop map to use in workshop) 1-2 days

b. Expert workshops / consultations (get experts together, talk about methods and principles of causal mapping, practice on software in preparation for further work after workshop) 2 days

c. Aggregation of models 2 days

d. Management of expert feedback 2 days

e. Write-up 1-2 weeks

**7.** Bayesian Network development

a. Pre-consultation preparation 2-4 days

b. Expert workshops / consultations for structure of network (first iteration, produces unparameterized causal network) 1 week

c. Parameter estimation

Elicitation from experts 1 week

Learning parameters from data (creating spreadsheet to use in conjunction with Netica) 2 weeks

d. Sensitivity analysis 3 days

e. Review 2 days

The time required to develop a BN varies according to the complexity and scale of the system. Two examples are presented below, a reasonably simple BN developed by Walshe and Massenbauer (2008) developed to aid decision-making for a Ramsar-listed wetland in W.A. The other example is a more complex BN developed to help manage Swamp Gum dieback in the Yellingbo Conservation Reserve (Pollino et al. 2007).

**Example One: Ramsar wetland (Walshe and Massenbauer 2008)**

BN with 11 nodes and 20 links. The structure of the model and parameter estimation was developed from the collaboration between the model developer and a domain expert. The parameters were estimated using expert knowledge. Mapping out the structure of the BN took about one day; filling in the CPTs (parameter estimation) took about one and a half days. The expert then spent one day with three other experts getting feedback about the model.

a) Pre-consultation preparation 2 days

b) Expert consultation (one expert) 1 day

c) Parameter estimation (one expert) 1.5 days

d) Review (three experts) 1 day

**Example Two: Yellingbo (Pollino et al. 2007)**

BN with 45 nodes and 77 links. A comprehensive list of threats and the most appropriate management endpoint (dieback in Swamp Gums) were identified from a workshop involving a wide range of interest groups. The structure of the BN was developed using the literature and the workshop proceedings. The model was parameterized using hydrological data, and information and data from the University of Melbourne (comparing leaf and water chemistry with level of dieback in trees), ecological data from a consultancy report (level of dieback in trees), and expert opinion.

| | | |
|---|---|---|
| a) | Pre-consultation preparation | 2 days |
| b) | Expert consultation (workshop) | 2 days |
| c) | Literature review | 4 weeks |
| d) | Structure developed | 1 week |
| e) | Parameter estimation (literature, data) | 4 weeks |
| f) | Analysis (model learning, sensitivity analysis) | 2 weeks |
| g) | Report | 2 weeks |

The development of a complex, large scale BN will take between three and six months, depending on the scale and complexity of the system, and especially on dynamic components of the system, which can be hard to correctly represent in a BN, and also whether the question that the BN is addressing has been sufficiently well defined.

Most of the time and effort is in getting the structure of the model right. Once this is developed to a level where the stakeholders are satisfied with the model structure the parameters can be estimated. The model is then presented to stakeholders to get input, using several subject area experts (for different areas of the same model) seems to work well. This gives the stakeholders something to react to, encourages feedback, and allows refinement of the CPTs. Eliciting CPTs directly is not generally required. Instead the experts are given the best case and worst case for each CPT as well as an explanation of the rules that are built into the model. If there is still major disagreement a questionnaire can be used to elicit further feedback.

## 3.8. Software

The software used to develop the casual map (Section 3.2) was CmapTools (version 4.16). This software was developed by the Institute for Human and Machine Cognition (IHMC 2008) by a collaboration of Florida University researchers (http://cmap.ihmc.us/conceptmap.html). This software is free and can be used for commercial or non-commercial purposes.

The Fuzzy Cognitive Map (Section 3.3) was developed using Hot Fuzz. This software was developed by the Department of Computer Science and Software Engineering at the University of Melbourne. This software is free and available at http://www.acera.unimelb.edu.au/resources.html (Cognitive Mapping Free Software) as well as a user's manual. Hot Fuzz supports semi-quantitative inference.

Netica was used to develop the Bayesian Network for grassland condition (www.Norsys.com). It coats $1,200 per annum for a site license for Netica, which allows the organisation to download the software on an unlimited number of work stations. It is supported by a help desk accessed through their website. Netica supports quantitative inference.

The State-Transition models were developed using PowerPoint.

# 4. DISCUSSION

Ecological interactions are complex and conservation management problems are typically ill-defined and poorly structured. The complexity of natural systems do not lend themselves to structuring and formulation by elaborate quantitative models, or simple intuitive problem solving. Rather, making sense of these situations necessitates considering, and often times negotiating, alternative models of the ill-structured situation. Graphical capture of individual and collective narratives of cause and effect assist problem formulation by facilitating the sharing of alternative perspectives and working towards a collective perspective (Massey and Wallace 1996).

Conceptual models are able to formally represent a summary of expert understanding about ecosystems, and be used to identify and prioritize specific information needs associated ecosystem management. In this way conceptual models form foundation for an integrated and holistic approach to management and research. The modeling methods all represent system features in a slightly different way, because of the characteristics inherent in each method.

### Strengths in all methods:

- As all methods are diagrammatic they can all be used to facilitate stakeholder engagement by providing a framework for dialogue.
- The structure of the conceptual models may be derived from any number of sources, including literature searches, expert elicitation, and qualitative or quantitative data.
- All of these methods can be used to decompose or partition complex systems into solvable steps.
- These modeling methods, rather than aiming to find definitive solutions to ecological management questions, can be used to promote further learning about a problem and the consequences of adopting a particular management option (Montibeller and Belton 2006).

### Weaknesses in all methods:

- These methods are generally not good at representing process complexity in great detail; they are most useful where the relationships of cause and effect are enough to capture the way the system functions (Cain 2001, Stow et al. 2003).
- Model structure is another source of uncertainty (Borsuk et al. 2004), which arises from different understandings of cause and effect, and the way in which the system functions (Burgman 2005). Hence it is an important part of conceptual model methodology to elicit information and feedback regarding model structure.

**Things to consider:**

Societal and personal values can not be separated from measures of condition for natural systems (Borsuk et al. 2003). The modelling methods used in this project could be used to incorporate these values by developing decision criteria that is values based rather than belief based, thus characterizing societal desires rather than behaviour of the natural system. These values, which would be reflected in the preferred state of key variables, can be elicited from stakeholders (the ‘desired state’ that forms the basis of management objectives). Related issues that require elicitation are what values managers should be trying to protect and what the management objectives should include. This would convert the conceptual models into decision models. Undergoing this process will help to minimise uncertainty in causal predictions and to ensure clarity about whether the origins of stakeholder disagreement are from differences in causal beliefs or goal preferences (Maguire 2004).

The hypotheses, assertions, literature and observations which justify the structure of each model should be fully documented. This allows the mechanistic foundation of models to be evaluated and challenged by the scientists and management communities, and allows model structures to evolve.

Pegler (2009) proposed that multi-criteria analysis (MCA) techniques be used to provide structured, transparent, and objective support to multi-stakeholder environmental decisions (Hajkowicz 2008). These techniques can be integrated with causal modelling in an adaptive management framework such that the predicted performance of decision options can be assessed against the full range of management objectives considering their relative importance to stakeholders (Linkov et al. 2006).

# 5. CONCLUSIONS

The management of natural values in Victoria's parks would benefit from a comprehensive causal model for each ecosystem type, which would include all threats, values, important processes and potential measurables for monitoring purposes.

It was identified in the introduction to this report that an ideal modelling approach would be one that (a) effectively captures ecological interactions; (b) is simple enough for operational use; (c) communicates causal understanding effectively to managers and stakeholders; and (d) is not prohibitively expensive in the time and resources required for model construction. The BN approach is probably the method best suited to achieve the first three of these aims. It has a major advantage in that it may also be used to explore the effect of different management and climate scenarios on system condition, and provide immediate feedback for different management actions. Netica is reasonably affordable, but the BN modelling method is prohibitive in the amount of time required to sufficiently parameterize even a moderately complex network. A good compromise would be the use of a causal map as the comprehensive and overarching framework for each ecosystem group, and the development of ST models that include management alternatives, as part of the model hierarchy.

Bayesian networks would remain a possibility for use on specific management issues, where the management problem is complex and their may be diverse understandings of causality and the impacts of intervention, and perhaps a need to develop a common understanding amongst stakeholders. BNs can be subsequently converted to decision networks by the addition of utility nodes which combine likely outcomes with social preferences. We could also incorporate other decision analysis approaches such as multi-criteria decision analysis. Fuzzy Cognitive Maps may be used to aid in the elicitation process.

The development of the causal models would include a process of stakeholder consultation, in the elicitation of threats and values, and also in ascertaining the causal structure of the system. Because of time constraints the models presented in this report were developed mainly from the literature, with expert input sought when the models were well-developed. It would be preferable to engage a large group of stakeholders at the earliest stage possible (after basic causal models have been drawn up to provide a framework for elicitation). Attached to the casual model (as part of the model hierarchy, see Section 3.2) would be the basic state-transition models, which would give more detail than is possible in the causal model. These would detail the likely consequences of different management actions (or the absence of specific actions).

Natural systems are complex, with many interacting components and many potential responses to management actions. It is difficult for managers and individual experts to conceptualise these systems, and therefore to make decisions regarding their management.

The information required to make sensible decisions about ecosystem management is commonly fragmented and diffuse. Currently, the information required to manage parks resides in branch and regional offices, internal reports, peer-reviewed literature, unpublished data and the knowledge of experts and other external stakeholders. Ecosystem models have the potential to bring this information and knowledge together as an integrated whole, identifying threats to the biological values of the parks, the causal structure of ecosystems and the likely outcomes of specific management interventions. They will also promote understanding and support communication within PV and with external stakeholders, by providing a transparent way to communicate the rationale behind management actions. The models will also help to build consensus, and where this is not possible, represent areas of disagreement.

Addressing objectives associated with the management of natural systems cannot be restricted by incomplete or biased empirical information, or bounded by spatial or temporal constraints typical of empirical information (McNay et al. 2006). Decisions about management will be made by managers even when faced with uncertainty. The aim of this report was to investigate methods for using the information available (from all sources) to make clear, explainable management decisions, and identify areas for further investigation.

# 6. ACKNOWLEDGEMENTS

There were many people involved in the work contained in this report. We would particularly like to thank Tony Varcoe, John Wright, Phil Pegler, Mark Antos, Marie Keatley, Steffan Howe, Fiona Smith and Brendan Sullivan for their time in providing technical knowledge, advice and feedback throughout the model development process. I would also like to thank Terry Walshe and Mick McCarthy from the University of Melbourne for their input and support in regards to modelling of ecological systems.

# 7. REFERENCES

Allen-Diaz, B., and Bartolome, J. W. (1998). Sagebrush-grass vegetation dynamics: comparing classical and state-transition models. *Ecological Applications* 8, 795-804.

Axelrod, R. (1976). *Structure of Decision: The Cognitive Maps of Political Elites*. Princeton University Press, New Jersey.

Barton, D. N., Saloranta, T., Moe, S. J., Eggestad, H. O., and Kuikka, S. (2008). Bayesian belief networks as a meta-modelling tool in integrated river basin management - Pros and cons in evaluating nutrient abatement decisions under uncertainty in a Norwegian river basin. *Ecological Economics* 66, 91-104.

Batchelor, C., and Cain, J. (1999). Application of belief networks to water management studies. *Agricultural Water Management* 40, 51-57.

Bestelmeyer, B. T., Brown, J. R., Havstad, K. M., Alexander, R., Chavez, G., and Herrick, J. E. (2003). Development and use of state-transition models for rangelands. *Journal of Range Management* 56, 114-126.

Borsuk, M. E. (2004). Predictive assessment of fish health and fish kills in the Neuse River estuary using elicited expert judgement. *Human and Ecological Risk Assessment* 10, 415-434.

Borsuk, M. E., Stow, C. A., and Reckhow, K. H. (2003). Integrated approach to total maximum daily load development for Neuse River Estuary using Bayesian probability network model (Neu-BERN). *Journal of Water Resources Planning and Management* July/August, 271-282.

Bowker, M. A., Belnap, J., Davidson, D. W., and Goldstein, H. (2006). Correlates of biological soil crust abundance across a continuum of spatial scales: support for a hierarchical conceptual model. *Journal of Applied Ecology* 43, 152-163.

Briske, D. D., Fuhlendorf, D., and Smeins, F. E. (2003). Vegetation dynamics on rangelands: a critique of current paradigms. *Journal of Applied Ecology* 40, 601-614.

Burgman, M. A. (2005). *Risks and decisions for conservation and environmental management*. Cambridge University Press, Melbourne.

Cain, J. (2001). *Planning improvements in natural resources management: Guidelines for using Bayesian networks to support the planning and management of development programmes in the water sector and beyond*. Centre for Ecology & Hydrology, Oxon, UK.

Cain, J., Batchelor, C., and Waughray, D. (1999). Belief networks: a framework for the participatory development of natural resource management strategies. *Environment, Development and Sustainability* 1, 123-133.

Cole, I., and Lunt, I. D. (2005). Restoring Kangaroo Grass (*Themeda triandra*) to grassland and woodland understoreys: a review of establishment requirements and restoration exercises in south-eastern Australia. *Ecological Management and Restoration* 6, 28-33.

de Bruin, W. B., Guvenc, U., Fischoff, B., Armstrong, C. M., and Caruso, D. (*In press*). Communicating about xenotransplantation: models and scenarios. *Risk Analysis*

Diez, S., and Foreman, P. (1997). *Remnant vegetation survey and botanical inventory of the Loddon Shire (part2) (Former Shires of East Loddon, Korong and the former City of Marong)*. Unpublished report to Loddon Shire Council and Natural Resources & Environment.

Dorrough, J., Yen, A., Turner, V., Clark, S. G., Crosthwaite, J., and Hirth, J. R. (2004). Livestock grazing management and biodiversity conservation in Australian temperate grassy landscapes. *Australian Journal of Agricultural Research* 55, 279-295.

Eden, C. (2004). Analyzing cognitive maps to help structure issues or problems. *European Journal of Operational Research* 159, 673-686.

Eleye-Datubo, A. G., Wall, A., Saajedi, A., and Wang, J. (2006). Enabling a powerful marine and offshore decision-support solution through Bayesian network technique. *Risk Analysis* 26, 695-721.

Grigg, G. (2002). Conservation benefit from harvesting kangaroos: status report at the start of a new millennium. In *A zoological revolution. Using native fauna to assist in its own survival* (Lunney, D., and Dickman, C., eds.), pp. 53-76. Australian Museum, Mosman.

Hajkowicz, S. A. (2008). Supporting multi-stakeholder environmental decisions. *Journal of Environmental Management* 88, 607-614.

Hart, B. T., Burgman, M., Grace, M., Pollino, C. A., Thomas, C., and Webb, J. A. (2006). Risk-based approaches to managing contaminants in catchments. *Human and Ecological Risk Assessment* 12, 66-73.

Heckerman, D. (1997). Bayesian Networks for data mining. *Data Mining and Knowledge Discovery* 1, 79-119.

Hobbs, B. F., Ludsin, S. A., Knight, R. L., Ryan, P. A., Biberhofer, J., and Ciborowski, J. J. H. (2002). Fuzzy cognitive mapping as a tool to define management objectives for complex ecosystems. *Ecological Applications* 12, 1548-1565.

Holling, C. S. (1973). Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics* 4, 1-23.

Holling, C. S., and Allen, C. R. (2002). Adaptive inference for distinguishing credible from incredible patterns in nature. *Ecosystems* 5, 319-328.

Huff, A. S. (1990). *Mapping strategic thought*. Wiley, New York.

Jackson, R. D., Bartolome, J. W., and Allen-Diaz, B. (2002). State and transition models: response to an ESA symposium. *Bulletin of the Ecological Society of America* 83, 194-196.

Klein, J. H., and Cooper, D. F. (1982). Cognitive maps of decision-makers in a complex game. *Journal of the Operational Research Society* 33, 63-71.

Kok, K. (2009). The potential of fuzzy cognitive maps for semi-quantitative scenario development, with an example from Brazil. *Global Environmental Change* 19, 122-133.

Kosko, B. (1986). Fuzzy cognitive maps. *International Journal of Man-Machine Studies* 24, 65-75.

Laycock, W. A. (1991). Stable states and thresholds of range condition on North American rangelands: a viewpoint. *Journal of Range Management* 44, 427-433.

Levin, S. A. (1992). The problem of pattern and scale in ecology. *Ecology* 73, 1943-1967.

Levins, R. (1966). The strategy of model building in population biology. *American Scientist* 54, 421-431.

Linkov, I., Satterstrom, F. K., Kiker, G., Batchelor, C., Bridges, T., and Ferguson, E. (2006). From comparative risk assessment to multi-criteria decision analysis and adaptive management: recent developments and applications. *Environment International* 32, 1072-1093.

Lunt, I. D., Eldridge, D. J., Morgan, J. W., and Witt, G. B. (2007). A framework to predict the effects of livestock grazing and grazing exclusion on conservation values in natural ecosystems in Australia. *Australian Journal of Botany* 55, 401-415.

Lunt, I. D., and Morgan, J. W. (1999). Vegetation changes after 10 years of grazing exclusion and intermittent burning in a *Themeda triandra* (Poaceae) grassland reserve in South-eastern Australia. *Australian Journal of Botany* 47, 537-552.

Maguire, L. A. (2004). What can decision analysis do for invasive species management? *Risk Analysis* 24, 859-868.

Marcot, B. G., Steventon, J. D., Sutherland, G. D., and McCann, R. K. (2006). Guidelines for developing and updating Bayesian belief networks applied to ecological modeling and conservation. *Canadian Journal of Forest Research* 36, 3063-3074.

Massey, A. P., and Wallace, W. A. (1996). Understanding and facilitating group problem structuring and formulation: Mental representations, interaction, and representation aids. *Decision Support Systems* 17, 253-274.

McIntyre, S., and Tongway, D. (2005). Grassland structure in native pastures: links to soil surface condition. *Ecological Management and Restoration* 6, 43-50.

McNay, R. S., Marcot, B. G., Brumovsky, V., and Ellis, R. (2006). A Bayesian approach to evaluating habitat for woodland caribou in north-central British Columbia. *Canadian Journal of Forestry Research* 36, 3117-3133.

Mingers, J., and Rosenhead, J. (2004). Problem structuring methods in action. *European Journal of Operational Research* 152, 530-554.

Montibeller, G., and Belton, V. (2006). Causal maps and the evaluation of decision options - a review. *Journal of the Operational Research Society* 57, 779-791.

Morgan, J. W., and Lunt, I. D. (1999). Effects of time-since-fire on the tussock dynamics of a dominant grass (*Themeda triandra*) in a temperate Australian grassland. *Biological Conservation* 88, 379-386.

Morgan, M. G., Pitelka, L. F., and Shevliakova, E. (2001). Elicitation of expert judgements of climate change impacts on forest ecosystems. *Climatic Change* 49, 279-307.

Nadkarni, S., and Shenoy, P. P. (2001). A Bayesian network approach to making inferences in causal maps. *European Journal of Operational Research* 128, 479-498.

Neuhauser, C. (2001). Mathematical challenges in spatial ecology. *Notices of the AMS* 48, 1304-1314.

Nyberg, J. B., Marcot, B. G., and Sulyma, R. (2006). Using Bayesian belief networks in adaptive management. *Canadian Journal of Forest Research* 36, 3101-3116.

Pearl, J. (2000). *Causality: models, reasoning, and inference*. Cambridge University Press, Cambridge.

Pegler, P. (2009). *The use of causal maps to support decision-making in environmental management*. University of Melbourne, Parkville.

Pellant, M., Shaver, P. L., Pyke, D. A., and Herrick, J. E. (2005). *Interpreting indicators of rangeland health, Version 4. Technical Reference 1734-6*. U.S. Department of the Interior, Bureau of Land Management, National Science and Technology Center, Denver, Colorado.

Pollino, C. A., White, A. K., and Hart, B. T. (2007). Examination of conflicts and improved strategies for the management of an endangered Eucalypt species using Bayesian networks. *Ecological Modelling* 201, 37-59.

Prober, S. M., and Thiele, K. R. (2005). Restoring Australia's temperate grasslands and grassy woodlands: integrating function and diversity. *Ecological Management and Restoration* 6, 16-27.

Prober, S. M., Thiele, K. R., and Lunt, I. D. (2002b). Identifying ecological barriers to restoration in temperate grassy woodlands: soil changes associated with different degradation states. *Australian Journal of Botany* 50, 699-712.

Ramsey, D. S. L., and Norbury, G. L. (2009). Predicting the unexpected: using a qualitative model of a New Zealand dryland ecosystem to anticipate pest management outcomes. *Austral Ecology* 34, 409-421.

Reckhow, K. H. (1999a). Water quality prediction and probability network models. *Canadian Journal of Fisheries and Aquatic Science* 56, 1150-1158.

Reckhow, K. H. (1999b). Lessons from risk assessment. *Human and Ecological Risk Assessment* 5, 245-253.

Reichert, P., Borsuk, M. E., Hostmann, M., Schweizer, S., Sporri, C., Tockner, K., and Truffer, B. (2005). Concepts of decision support for river rehabilitation. *Environmental Modelling and Software*??, 1-14.

Rogers, K., and Biggs, H. (1999). Integrating indicators, endpoints and value systems in strategic mangement of the rivers of the Kruger National Park. *Freshwater Biology* 41, 439-451.

Rushton, S. P., Luff, M. L., and Eyre, M. D. (1989). Effects of pasture improvement and management on the ground beetle and spider communities of upland grasslands. *Journal of Applied Ecology* 26, 489-503.

Stow, C. A., Roessler, C., Borsuk, M. E., Bowen, J. D., and Reckhow, K. H. (2003). Comparison of estuarine water quality models for total maximum daily load development in Neuse River estuary. *Journal of Water Resources Planning and Management* July/August, 307-314.

Stringham, T. K., Krueger, W. C., and Shaver, P. L. (2003). State and transition modeling: an ecological process approach. *Journal of Range Management* 56, 106-113.

Stylios, C. D., Georgopoulos, V. C., and Groumpos, P. P. (Year). "The use of fuzzy cognitive maps in modeling systems." Paper presented at the 5th IEEE Mediterranean Conference on Control and Systems, Paphos, Cyprus, July, 1997.

Suter, G. W. (1995). Introduction to ecological risk assessment for aquatic toxic effects. In *Fundamentals of aquatic toxicology: Effects, environmental fate and risk assessment*, 2nd ed. (Rand, G. M., ed. Taylor and Francis, Washington.

Verrier, F. J., and Kirkpatrick, J. B. (2005). Frequent mowing is better than grazing for the conservation value of lowland tussock grassland at Pontville, Tasmania. *Austral Ecology* 74-78.

Walker, B. (1995). Conserving biological diversity through ecosystem resilience. *Conservation Biology* 9, 747-752.

Walshe, T., and Massenbauer, T. (2008). Decision-making under climatic uncertainty: a case study involving an Australian Ramsar-listed wetland. *Ecological Management and Restoration* 9, 202-208.

Walters, C. J., and Holling, C. S. (1990). Large-scale management experiments and learning by doing. *Ecology* 71, 2060-2068.

Westoby, M., Walker, B., and Noy-Meir, I. (1989). Opportunistic management for rangelands not at equilibrium. *Journal of Range Management* 42, 266-274.

Wilhere, G. F. (2002). Adaptive management in habitat conservation plan. *Conservation Biology* 16, 20-29.

Wilkinson, S. R., Naeth, M. A., and Schmiegelow, F. K. A. (2005). Tropical forest restoration within Galapagos National Park: application of a state-transition model. *Ecology and Society* 10, 1-16.

Wong, N., and Morgan, J. W. (2007). *Review of grassland management in south-eastern Australia*. Parks Victoria Technical Report No. 39. Parks Victoria, Melbourne.

Yen, A. L. (Year). "Grassland invertebrates of the Western Victorian Basalt Plains: plant crunchers or forgotten lunches?" Paper presented at The Great Plains Crash Grasslands Conference, Victorian University of Technology, Footscray, 1992.

*Parks Victoria is responsible for managing the Victorian protected area network, which ranges from wilderness areas to metropolitan parks and includes both marine and terrestrial components. Our role is to protect the natural and cultural values of the parks and other assets we manage, while providing a great range of outdoor opportunities for all Victorians and visitors.*

*A broad range of environmental research and monitoring activities supported by Parks Victoria provides information to enhance park management decisions. This Technical Series highlights some of the environmental research and monitoring activities done within Victoria's protected area network.*

*Healthy Parks Healthy People*

For more information contact the **Parks Victoria Information Centre** on **13 1963**, or visit **www.parkweb.vic.gov.au**

Victoria The Place To Be

*Healthy Parks Healthy People*

Parks VICTORIA