Department of Sustainability and Environment

Review of resilience concepts and their measurement for fire management

Fire and adaptive management

report no. 90

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Front cover image
Eucalypt regeneration after fire 2009, G. FRIEND.

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Contents

Introduction

There is increasing focus from researchers and land managers on the stability and diversity of social and ecological systems, the resilience of these systems to change, and the ability to manage these systems (Holling 1973, Gunderson 2002). These ideas may be particularly applicable to complex systems where managers must make decisions in the presence of incomplete information (Peterson et al. 2003). Adaptive management is a parallel approach to decision making when knowledge is limited (Holling 1978, Walters 1986, Williams 2001, McCarthy and Possingham 2007). I review these two related approaches to considering management, with particular attention to developing appropriate fire management through use of prescribed fire and fire suppression activities.

This review arises from a project associated with the Future Fire Management Project of the Victorian Department of Sustainability and Environment (DSE). The following points were provided by DSE as background, and set the theme for the project.

1. Risk and Adaptive Management (RAM) requires the use of available knowledge to assess the current status of risk and natural resources, predict possible future outcomes, and set objectives and strategies while at the same time acknowledging the uncertainty involved. Resulting plans, policies and practices are altered as knowledge from research, monitoring and organisational and individual learning and expectations/values change.
2. Adopting the principles, practice and discipline involved in RAM enables progress in the absence of complete knowledge, because managed experimentation and learning is not only acceptable but encouraged.
3. Applying RAM principles also allows land and bushfire managers, stakeholders and interested communities to focus on strategic discussions, understanding what is known, and differing values and expectations in working towards improved agreement.
4. Resilience has been extensively discussed in academic and related literature. However, general objectives and strategies aimed at improving organisational, social, economic and environmental resilience, and general measures of resilience, are not readily available.
5. For land and bushfire management, the long timeframes and scales that policy inputs, actions and eventual outcomes operate over present a major challenge when assessing the validity and effectiveness of strategies and actions.
6. Long-term strategies for bushfire management need to incorporate risk and adaptive management, work towards objectives and outcomes that relate to improved resilience, and incorporate realistic measures of environmental impacts. These are a necessary first step in achieving outcomes to better manage fire in an inherently fire-prone environment in which risks (e.g. from climate change, further rural development and increased community expectations) are expected to increase.
7. This project aims to provide a concise description of resilience and to develop working principles and guidance on how to frame objectives, outcomes and measures of resilience from an environmental perspective. These principles and measures will feed into the development of initial landscape-scale fire management options ('regimes'), as part of the broader Future Fire Management Project.

This review seeks to:

1. Define resilience and associated thinking and how it may be applied to environmental objectives – supported by example where possible.
2. Describe metrics of 'states' of resilience.
3. Develop working principles and guidance on how to frame objectives, outcomes and measures of resilience from an environmental perspective.
4. Apply these ideas and metrics for the development of initial landscape-scale fire management options ('regimes'), as part of the broader Future Fire Management Project.

The review initially examines questions of stability and diversity as background for discussing resilience and related concepts. The review examines how stability, diversity and resilience can be measured, asks how they can be predicted *a priori,* and examines their application to fire management. Adaptive management is presented as a possible approach to managing the vagaries and uncertainties associated with the resilience of social and ecological systems. The concepts developed in the review are applied to different aspects of the Otways case study area.

Stability, diversity and resilience

Stability and diversity

The relationships between stability and diversity have been an enduring topic in ecology. Darwin touched on the issue in his *Origin of Species* (1859, chapter 4, 'Divergence of Character'). Elton (1958) was one of the first modern ecologists to suggest that diversity begets stability, a hypothesis based on observation. For example, species-poor vegetation types such as agricultural fields seemed more susceptible to invasive plant species than natural vegetation communities that contained more species. Elton believed that more complex communities were less sensitive to changes in the abundance of any one predator or parasite. These ideas reflected MacArthur's (1955) view that any change in the population size of a species when one of its prey or predators changed in abundance would be tempered if the species had a diversity of prey and predators.

Mathematical models developed by May (1973) challenged the intuition of ecologists that was reflected in the work of Elton (1958), MacArthur (1955) and others. May's models, and those of other ecologists, suggested that increased diversity would destabilise communities. The difference between theoretical predictions, intuition and observation set the stage for theoretical and empirical research on the question of whether and how diversity was related to stability (Yodzis 1981, Tilman 1996, Tilman et al. 1996, McCann 2000, Ives and Carpenter 2007).

When examining relationships between stability and diversity, the way that stability is measured is important. Stability can be measured by (Ives and Carpenter 2007):

1. whether a system returns to the same state (or dynamics for non-equilibrial systems) after a perturbation;
2. the speed at which a system returns to the same state (or dynamics for non-equilibrial systems) after a perturbation;
3. the variation over time in biomass or some other measure of a community;
4. the ability to withstand change (e.g. in terms of fluctuations, or changes between alternative states) caused by environmental perturbations;
5. the chance of a new invader becoming established, or a species becoming extinct; and
6. the number of extinctions caused by the extinction of one species in the community, or by the invasion of a new species.

This wide range of ways that stability can be measured presents an empirical challenge, especially when combined with the different ways of measuring diversity. As well as the basic measures of species diversity (e.g. number of species, diversity indices such as Shannon or Simpson diversity indices, etc.), functional diversity (e.g. the diversity of pollinators, nutrient cyclers, etc.) may be particularly important when examining relationships with stability (Walker et al. 1999).

Despite the empirical challenges of measuring stability and diversity, a positive relationship between the two is supported by mounting evidence, at least at relatively small spatial scales (Ives and Carpenter 2007). The principal mechanisms explaining the positive relationship are effects of (McCann 2000):

1. averaging – by averaging over uncorrelated fluctuations in all species, measures of variation in total community biomass, for example, can decline with increases in species richness (Doak et al. 1998);
2. negative covariance – if different species respond differently to environmental variation, some species will increase while others decline, reducing the variation in biomass or abundance of individuals, for example, of the total community (Tilman et al. 1998); and
3. insurance – functional redundancy, which may be increased with species diversity, may help to buffer a community from the effects of environmental perturbations (Naeem and Li 1997, Yachi and Loreau 1999).

However, it should be noted that correlation does not prove causation, so while diversity and stability may be correlated, there is only relatively limited empirical evidence that diversity increases stability directly.

The scale at which diversity and stability is measured is also important. Most empirical studies to date have focussed on the stability of relatively uniform areas (e.g. a single area of grassland, or a single lake). The relationships between stability and diversity at larger spatial scales (e.g. collections of interconnected lakes, or even compilations of elements such as lakes, grasslands and forest in a landscape) remain to be determined.

In considering the relationship between diversity and stability, Walker et al. (1999) note that functional diversity is regarded as being important for supporting the present flow of environmental goods and services (a form of stability). However, their novel suggestion is that functional similarity is important for maintaining the flow of environmental goods and services when a system changes. This is because a functional role of one species can be replaced by another, if the first declines in response to change. Essentially, they argue that stability is maintained by both differences and similarities among species. The buffering of functions brings us to the topic of resilience.

Resilience

Brand and Jax (2007) discuss various uses and definitions of the term resilience in environmental science, which are summarized briefly here. Resilience was initially defined as the capacity of ecological systems to absorb disturbance and change. This same basic concept has been used to consider the resilience of social systems and of economic-environmental systems. This led to resilience being defined as the capacity to maintain desirable ecosystem services, with consideration of explicit links between human and natural systems, and the resilience of social-ecological systems. The concept of resilience subsequently broadened as a way of conceptualising social-ecological systems, with resilience being a set of ideas for interpreting complex systems. Resilience has been further used to represent flexibility and sustainability. These later uses are difficult to define precisely in measurable terms, although Brand and Jax (2007) argue they are useful for communication across disciplines. The ecologically-focused definitions allow us to consider the resilience of specific attributes in the face of particular impacts, providing opportunities to define measurable aspects of resilience if those attributes and impacts are defined. In this report I focus on ecological aspects of resilience, with the aim of developing ways of measuring resilience of biodiversity in response to fire regimes.

Resilience is a form of stability, equivalent to the fourth definition provided above. While stability is often measured by the degree of variation in a community in its 'usual' state (which might be dynamic), resilience is related to the change in a community in response to a novel perturbation. Walker et al. (2004) define resilience as "the capacity of a system to absorb disturbance and reorganise while undergoing change so as to still retain essentially the same function, structure, identity and feedbacks".

Walker et al. (2004) identify four critical aspects of resilience (Figure 1):

1. precariousness – how close the system is to a fundamental change;
2. latitude – the degree of variability in the system while it still retains essentially the same function, structure, identity and feedbacks;
3. resistance – the effort required to change the system; and
4. panarchy – how the previous three aspects are influenced by the alternative systems into which the system may change, or the dynamics of a system within its current state.

For example, consider a wet sclerophyll forest in which the lifespan of individual eucalypt trees is 400 years, regeneration from seed requires fire, but seed is unavailable on trees until they are 20 years of age. Precariousness is in part measured by how close the fire interval is to being less than 20 years or greater than 400 years. A fire interval of less than

20 years would lead to the local extinction of the eucalypts, and the vegetation would perhaps be dominated by tall shrubs such as *Acacia* species. Rainforest species may replace the eucalypts as dominants if the fire interval were longer than 400 years. In any case, the system will have essentially changed without the eucalypts being present. Latitude is in part the range of fire intervals that allow the trees to regenerate. Latitude is also the range of variation in species diversity and forest structure that is experienced while the forest regenerates. Resistance is how difficult it is to extend the fire interval beyond 400 years, or to reduce it to less than 20 years. Panarchy in part refers to the attributes of the other systems (*Acacia* and rainforest communities), how resilient they are to change (e.g. through recolonisation by eucalypts following another fire) and how long an ecosystem takes to return to the initial condition or a similar state.

Spatial scale is also important when considering resilience. For example, an intense fire in an old-growth forest patch that is embedded within a much larger landscape of old forest will reduce resilience at the landscape scale less than if the burnt old-growth area is embedded within a landscape of young forest.

These concepts have been discussed in relation to the degree of change from one state to another. Perhaps implicit in this discussion is that the current state is desirable. However, the resilience of undesirable states can also be measured; we might aim to make undesirable states unresilient so that they can be changed more easily.

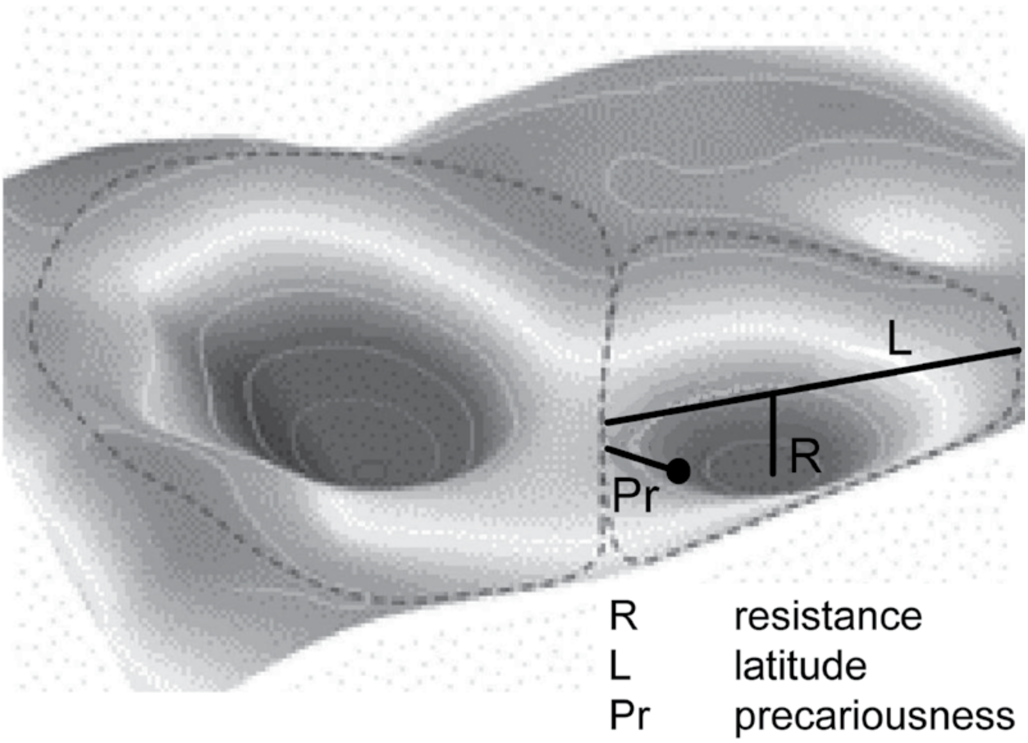

Fig. 1. A diagram illustrating resistance, latitude and precariousness, three aspects of resilience (from Walker et al. 2004). The landscape represents the environmental space, with each basin a particular state of the environment. A state is resistant to change if the basin is deep. Latitude measures the degree of variation across the environmental space that can be accommodated before the system changes to another state by entering an adjacent basin. Precariousness measures how close the current environment is to the edge of the basin – how close the system is to changing state by entering another basin. Panarchy integrates these three aspects across multiple basins. A state is made more resilient if the resistance, latitude and precariousness of the system when in adjacent basins facilitate change back to the original state.

A priori measurement of resilience

Measuring resilience is difficult in practice. Because it is the ability of an ecosystem to absorb disturbance and change without loss of function or structure, measuring it requires that the change in function or structure is observed. In many cases, maintaining the current state of an ecosystem is desirable, so measuring resilience by forcing a change is unacceptable. Unresilient changes are often obvious, but there are no proven general methods for predicting resilience *a priori.*

Peterson et al. (1998) note links between resilience and the diversity of body sizes of species. The basis of this idea is that landscapes with species across a wider range of body sizes (e.g. small, medium and large seed dispersers), and large numbers of species within each group, will be more resilient to disturbance. Peterson (2002) proposes that resilience of landscapes can be measured by cross-scale edge, providing a rough and rapid estimate of areas of resilience and vulnerability within a landscape. Allen et al. (2005) propose that functional groups and discontinuities between them can measure resilience (based on Peterson et al. 1998). The proposal of Allen et al. (2005) generalises the ideas of Peterson (2002) because vegetation types in a landscape can be considered equivalent to functional groups, and edge detection is equivalent to identification of discontinuities. For example, indices of resilience can be based on the number of functional groups (e.g. defined by feeding guild and substrate), the diversity of body masses (a surrogate for function given body mass is correlated with ecological rates and processes), and the number of species in each of these groups (e.g. Fischer et al. 2007). These and related indices represent a set of hypotheses for measuring resilience *a priori* (i.e. before it becomes obvious that a system is not resilient). There have been only a small number of relatively limited attempts to evaluate the performance of these indices, especially with regard to *a priori* prediction of resilience (Fischer et al. 2007, Wardwell et al. 2008).

Empirical research demonstrates that species richness contributes to maintaining ecosystem resilience (Yodzis 1981, Tilman 1996, Tilman et al. 1996, McCann 2000, Ives and Carpenter 2007). While loss of species would be a useful indicator of declining resilience, species richness may not necessarily be a good *a priori* predictor of resilience. Declines of species would help to indicate declining resilience prior to the loss of species, so indices that are sensitive to changes in the relative abundance of species may be useful measures of resilience. A recent microcosm experiment by Wittebolle et al. (2009) demonstrates that evenness of species abundances can impart resilience to environmental change. Evenness measures the similarity in the abundance of species within a community. For example, a community with three species and 300 individuals would have maximum evenness when there are 100 individuals of each species. The evenness of the community of three species would be at its minimum when there are 298 individuals of one species, and one individual of each of the other two. In all cases of a community of three species and 300 individuals, the arithmetic mean abundance of the species is the same (300/3 = 100). However, other averages, such as the geometric mean abundance, differ with evenness. When all species have 100 individuals, the geometric mean abundance is 100 ($(100 \times 100 \times 100)^{1/3}$), while it is at a minimum when there is only one individual of two species and 298 individuals of the third ($(1 \times 1 \times 298)^{1/3} = 6.68$). Indices that are sensitive to changes in the relative abundance of species, such as the geometric mean, may be useful indicators of resilience. Despite there being a range of possible indicators of resilience, no definitive studies demonstrate which measures to use in any particular circumstance. In this case, measures of resilience are required that are based on hypothesised relationships for a particular circumstance (see section **Measures of resilience based on modelling**), with monitoring and evaluation of these hypothesised relationships being necessary.

Application to landscape-scale fire management

The fire regime of a site is defined by four characters of fire events: the intensity of fires, the season in which they occur, the intervals between fires, and whether the fire burns surface fuels or below ground (Gill 1975). These four aspects of fire regimes can explain much of the ecological response of biodiversity to fires, especially when considering the variability in fire regimes, both in space and time (Russell-Smith et al. 1997, Gill and McCarthy 1998). Spatial aspects of fire regimes are important because species, in particular animals, might use resources from multiple locations that have experienced different fire regimes. Variability in fire regimes has been hypothesised to help maintain biodiversity (Keeley 1991, Gill and McCarthy 1998), and there is some evidence to support this (e.g. Jackson 1968, Keith and Bradstock 1994, Morrison et al. 1995).

In Jackson's study (1968), variability in fire regimes appears to support diversity as measured by the range of communities in the landscape (e.g. the mix of moorlands, rainforest, etc.). However, this study also demonstrates that the often-proposed relationship between diversity and stability can be case-dependent. For example, rainforests, a more species-diverse community, appears to be more likely to change state (i.e. become moorland) following fire. However, it is possible that more diverse rainforests will, on average, be more resilient to fire than less diverse rainforests.

Whereas natural systems are typically variable, and the diversity of these systems can depend on this variability, management often aims to limit and control this variability. McCarthy and Burgman (1995) show how variability in fire intervals influences the age structure of the landscape that results. For example, if the average fire interval at points in the landscape is 10 years, the average age of the landscape will be approximately half that interval when the fires occur regularly (e.g. due to prescribed fire, deterministic disturbance). In contrast, the average age will be approximately 10 years if the fires occur randomly (e.g. due to unplanned fire, stochastic disturbance) with the same average interval. The age structures of these two landscapes are very different (Figure 2) because stochastic disturbance allows some parts of the landscape to escape fire. In contrast, deterministic disturbance targets the oldest parts of the landscape (McCarthy and Burgman 1995).

The advantage of tightly controlled management is a relatively consistent supply of a narrow set of resources (e.g. timber products). However, this has the effect of reducing spatial and temporal variability (McCarthy and Burgman 1995), making change increasingly undesirable but more difficult to avoid (Holling 1986, Gunderson et al. 1995, Drever et al. 2006). This is an example of the "pathology of natural resource management" in which attempts at regulating services from natural resources reduce resilience and lead to failure of regulation (Holling and Meffe 1996). There is a reasonable body of evidence that diversity in biological composition and structure at a range of spatial scales contributes to different aspects of resilience (see material reviewed in Drever et al. 2006). Increasing the diversity of the age structure of a landscape (in terms of time since fire) and the diversity of fire intervals is consistent with these concepts of resilience. Diversity of intensity, season of occurrence and spatial structure is also consistent with these concepts. Increasing the probability that fire regimes lie within acceptable thresholds is an additional goal that would also help to promote resilience.

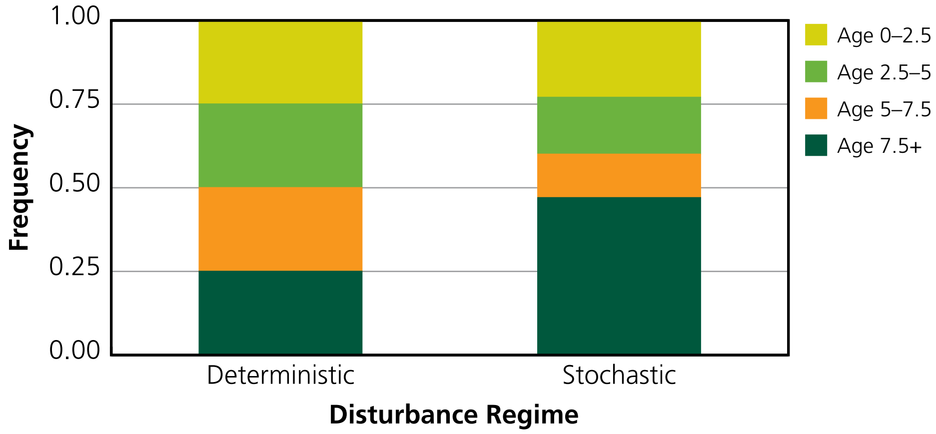

Fig. 2. The proportion of a landscape in each of four different age classes when disturbed by deterministic fires every 10 years (the oldest 10% of the landscape burning each year), or by stochastic (random) fires with an average interval for each point of the landscape of 10 years such that on average 10% of the landscape burns each year. Modified from McCarthy and Burgman (1995).

There is a trade-off, however, between some of these objectives. For example, fire intervals that are longer than a prescribed maximum can be readily avoided by using prescribed fire. However, avoidance of fire intervals shorter than a prescribed minimum rely on an absence of unplanned fire. Every time that a site is burnt makes a short fire interval possible. Therefore, the probability of a site being burnt by an unplanned fire with an undesirably short interval is minimised (ignoring unexpected escapes of planned fires) by delaying the application of fire until the site has reached its maximum tolerable age.

Decision theory methods provide tools for setting objectives and assisting managers to determine the most appropriate strategies for achieving these objectives. Richards et al. (1999) used decision theory methods to devise optimal fire management policies when the goal was to maintain a diversity of age classes in an area. They presented an example based on fire management in Ngarkat Conservation Park. The objective was set, for illustrative purposes, as maximising the expected number of years in which there was at least 20% of three different age classes. The management strategies were either to light fires in different age classes, fight fires to reduce the influence of unplanned fires, or do nothing. In the presence of unplanned fires, this results in a Markov decision process, the optimal management of which requires stochastic dynamic programming (Richards et al. 1999). The details of this method are not important for this review, except to say that stochastic dynamic programming identifies the strategy that best achieves the objective, and that this strategy is state-dependent; whether to light or fight fires depends on whether the landscape is dominated by early-, mid- or late-successional vegetation. The optimal strategy will also depend on the exact form of the objective function; not surprisingly, the solution depends on what the manager aims to achieve. It is also possible to incorporate costs of different management actions into the analysis. As an example of the results, and not surprisingly, lighting prescribed fires tends to be optimal when the youngest age class is underrepresented. Controlling fires is optimal when the oldest age class is underrepresented. However, active management to light or fight fires can be optimal even when the park is in the desired state. Further, when costs are considered, there are some conditions in which it is optimal to do nothing because the benefits of either fighting or lighting fires are small in these particular states (Figure 3).

As an example of applying the Richards et al. (1999) model to fire management in Victoria, here it is applied to foothills forest in the Otways. Data on the proportion of area burnt each year were fitted to a log-normal distribution (Figure 4) to define the proportion of the area burnt each year in which fires occurred. Early-successional vegetation was defined as

<10 years since the last fires, and late-successional vegetation was >25 years of age. It was assumed that the aim was to have at least 20% of the area in each of the three age classes (<10, 10–25, >25 years). Costs of fighting fires were ignored, and it was assumed that unplanned fires were fought. The results demonstrate how the optimal fire management policy depends on the previous fire history through its effect on age-structure of the landscape (Figure 5). Note that the goal of aiming to achieve at least 20% of each age class is arbitrary. A method to determine an appropriate age structure is described later in this report (see the section *Geometric mean abundance*).

These analyses were repeated for the other six most common vegetation types (Damp Scrub, Forby Forest, Heathland, Moist Forest, Tall Mist Forest and Tall Mixed Forest), which together with Foothills Forests represent the main ecological fire groups of the Otways study area (Figure 6). Times since fire were assigned to early-, mid- and late-successional stages (Table 1), using mapped data provided by DSE (Figures 6 and 7). When areas had no mapped fire history, it was assumed that the vegetation was late-successional, except in the case of Tall Mist Forest, where it was assigned to mid-successional.

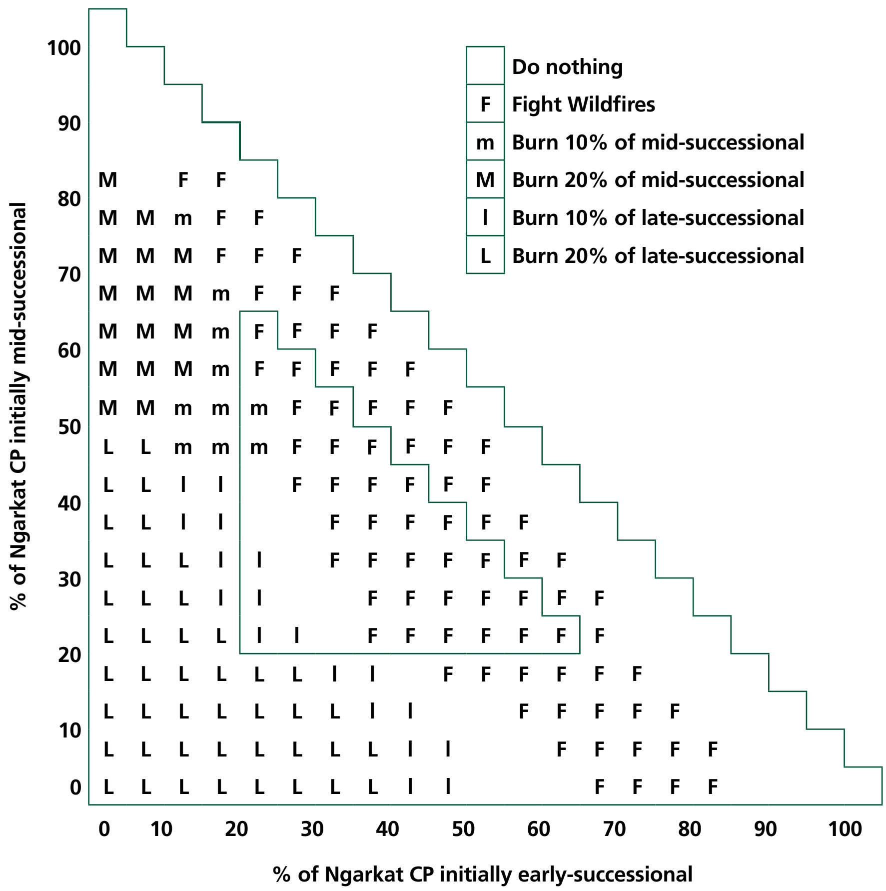

Fig. 3. An example of the optimal management strategy for managing age structure of vegetation in Ngarkat Conservation Park (Figure 5 from Richards et al. 1999). The triangular area in the middle represents the desired state. The symbols define the optimal strategy as a function of the amount of early- and mid-successional vegetation in the landscape, and, by implication, the remainder is late-successional vegetation.

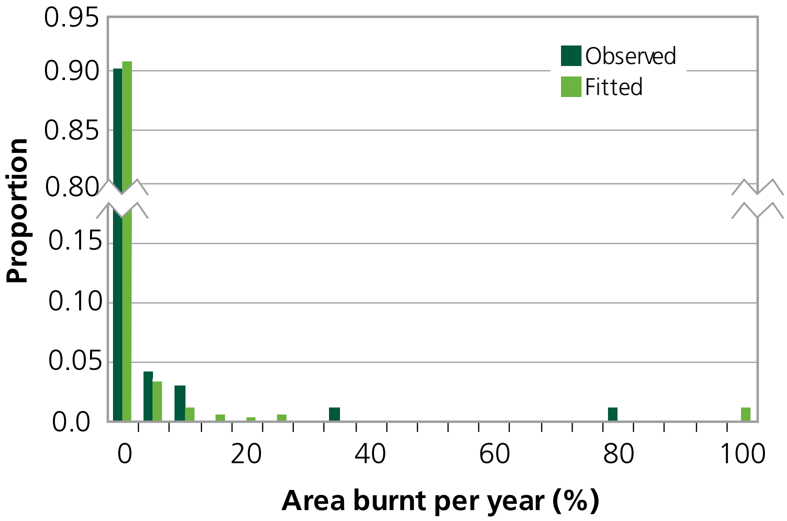

Fig. 4. Histogram of the distribution of the area of Foothills Forest burnt per year by unplanned fires in the Otways for the period 1939–2008, compared with a fitted distribution in which fires occur on average 2 years in 7, and the size of fires has a log-normal distribution in years when they do occur. The area of fires is pooled to the nearest 5% of the total area. The average proportion of the area burnt each year estimates the annual probability of fire.

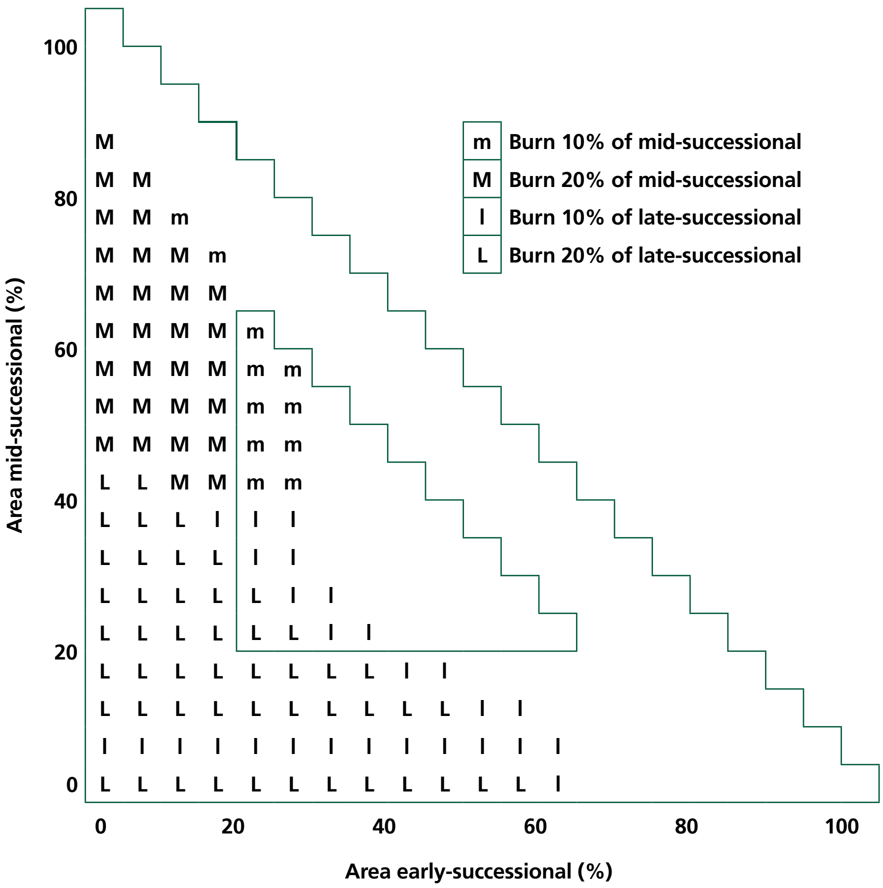

Fig. 5. The optimal fire management policy for the Foothills Forest of the Otways, assuming the distribution of unplanned fires given in Fig. 4, and with the aim of achieving at least 20% of the landscape in early-, mid- and late-successional stages, which are defined as <10, 10–25, >25 years since the last fires. The triangular area in the middle defines the desired distribution of age classes. The management time horizon was assumed to be 50 years.

Table 1. The range of ages (years since fire) used to define the mid-successional stage of the ecological fire groups used in this illustrative example; early- (late-) successional stages are younger (older) than these limits. The current distributions of age classes of the vegetation types in the Otways study area are also presented, along with the management option that maximises the probability of achieving an even distribution of ages given the current age structure.

Ecological Fire Group (EFG)	Age limits (mid)		Successional stage			Burn
	Lower	Upper	Early	Mid	Late	
Damp Scrub	15	50	0.08	0.49	0.43	20% mid
Foothills Forest	10	25	0.01	0.03	0.95	20% late
Forby Forest	15	40	0.13	0.27	0.60	20% late
Heathland (sands) – Default	12	30	0.14	0.43	0.43	10% mid
Moist Forest	25	60	0.04	0.28	0.67	20% late
Tall Mist Forest	80	200	0.45	0.55	0.00	–
Tall Mixed Forest (Eastern)	8	25	0.06	0.12	0.82	20% late

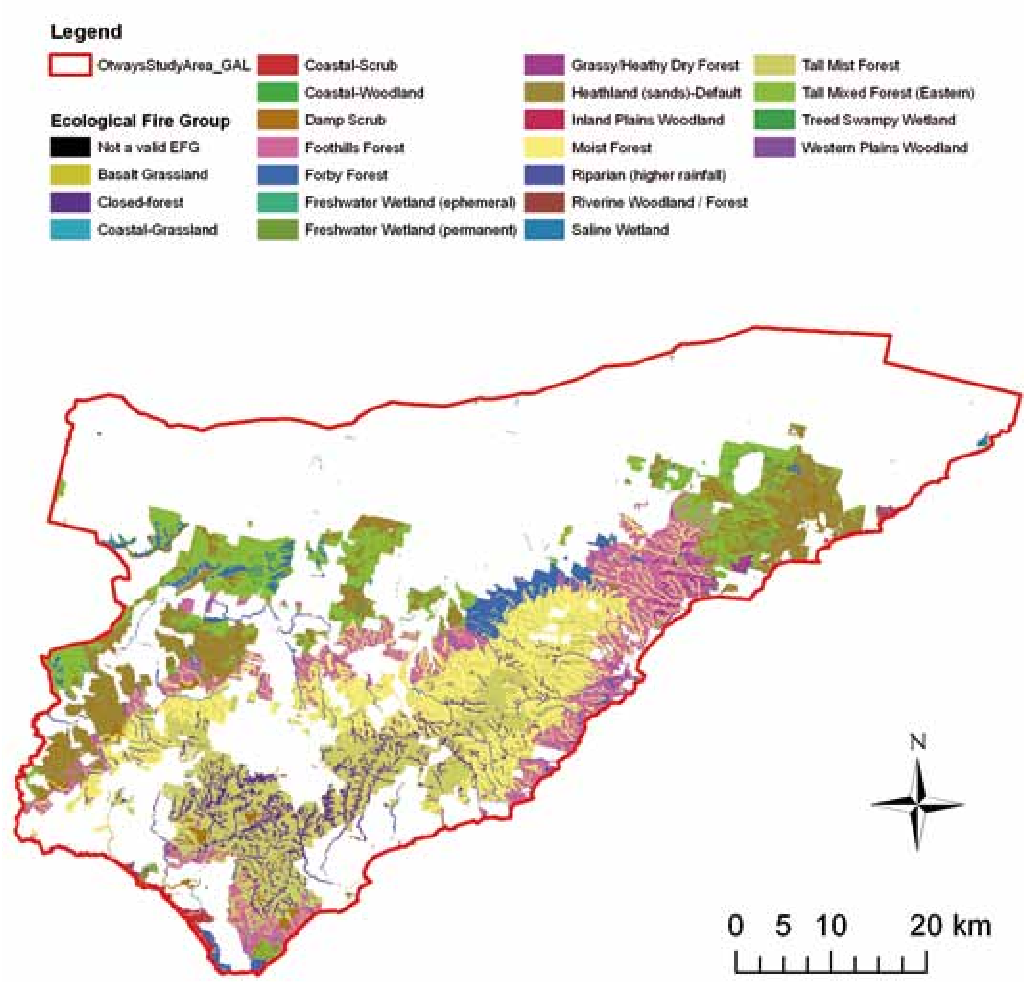

Fig. 6. Ecological Fire Groups in the Otways study area. Damp Scrub, Foothills Forest, Forby Forest, Heathland, Moist Forest, Tall Mist Forest and Tall Mixed Forest (Eastern) constitute almost all the native vegetation of the region. Spatial data provided by DSE.

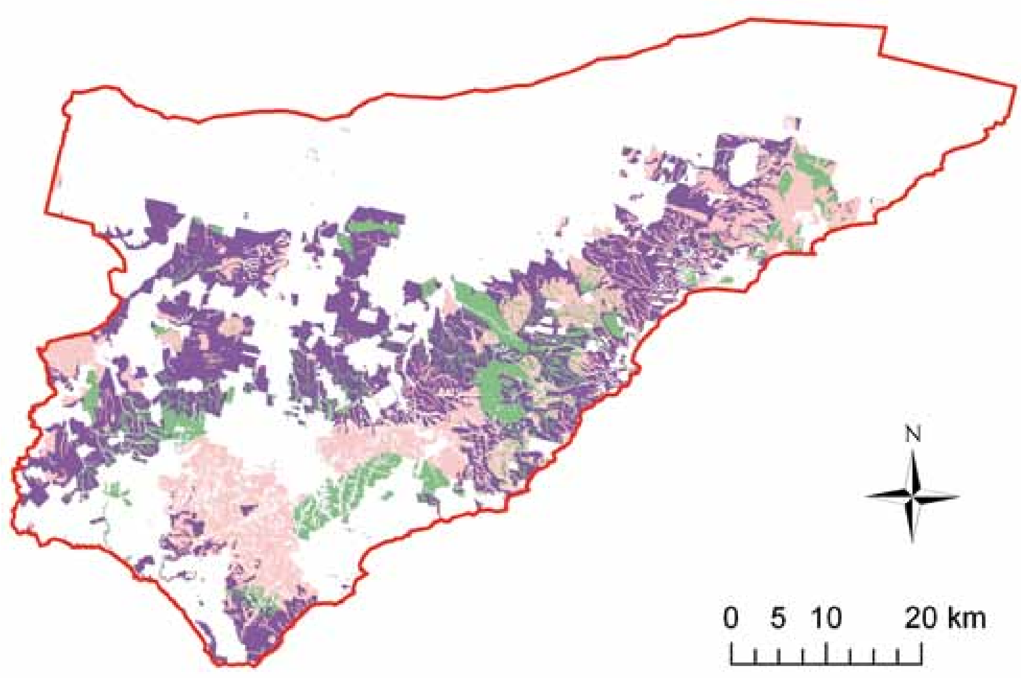

Fig. 7. The distribution of ages of the seven most common EFGs in the Otways study area classified into early- (green), mid- (pink) and late-successional (purple) vegetation based on the limits assumed in Table 1.

The optimal prescribed burning strategy can be determined for each of the EFGs based on the current distribution of successional stages. This is the strategy that, in the presence of unplanned fires, maximises the number of years within the next 50 for which the vegetation is expected to have at least 20% of each age class present. For example, the current high prevalence of late-successional vegetation in Foothills Forest means that burning 20% of this area is optimal (Table 1). In contrast, burning mid-successional Damp Scrub is optimal because this age class is over-represented (relative to the arbitrary goal of achieving at least 20% of each age class), and it is easier to regain this age class compared with late-successional vegetation in the presence of unplanned fires. In contrast, prescribed fire should not be applied to Tall Mist Forest given the current age structure. However, if the area with late-successional Tall Mist Forest was more than 25%, burning late-successional Tall Mist Forest would be optimal.

The spatial location of the vegetation types for which burning is optimal can be compared with the location of early-successional vegetation where fires should be excluded so that they do not burn at less than their tolerable fire intervals (Figure 8). This indicates areas where fire is preferred to occur, and where it is preferred to be excluded until the vegetation matures. The risk of prescribed burning near areas with early-successional vegetation needs to be weighed against the benefits in achieving the desired age structure. For example, burning late-successional Moist Forest might be risky given its proximity to mid-successional Tall Mist Forest, although prescribed burning in adjacent areas might help to manage the influence of fires in mid-successional Tall Mist Forest.

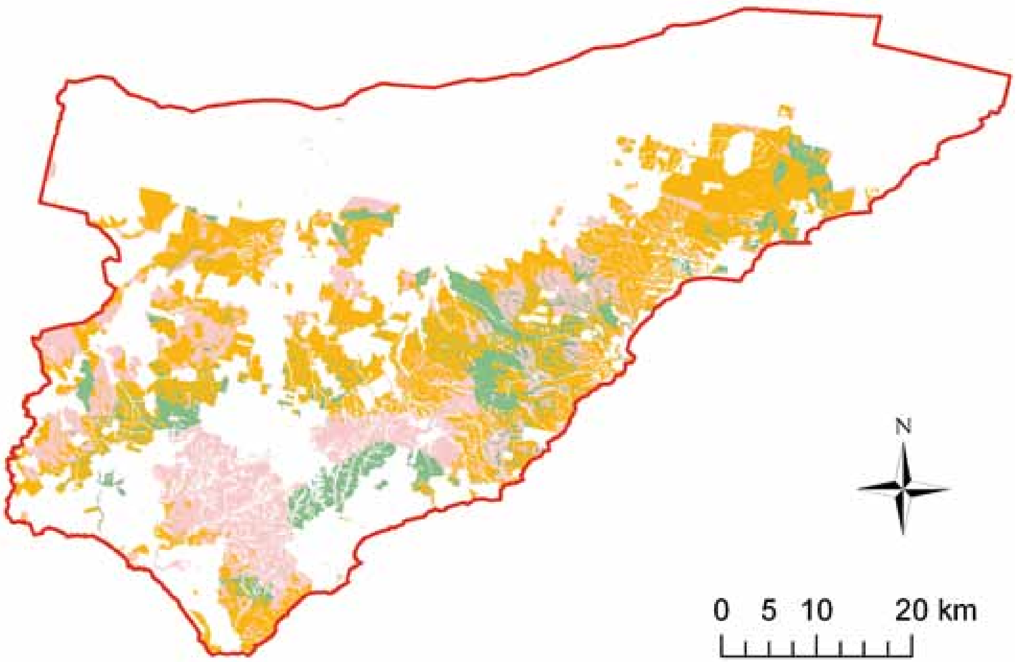

Fig. 8. The location of areas with age classes that require burning (orange) compared with early-successional vegetation (green) where burning should be avoided. Pink areas are the remaining areas of the seven EFGs in the case study.

The model of Richards et al. (1999) does not consider the season in which fires occur nor the intensity of the fires. Further, the model ignores the actual fire intervals at sites, instead summarising the state of the landscape by the age structure (the proportion of the landscape in each of the three age classes). In particular, the objective function could be modified to assess the trade-off between attaining a particular age structure and limiting the occurrence of short, unplanned fire intervals. Incorporating other aspects of the fire regime, such as season of occurrence and intensity, is conceptually easy, but may be computationally challenging. However, it is not clear that diversity in fire intensity and season of occurrence needs to be incorporated directly into the optimisation because they can probably be dealt with qualitatively, and, in any event, may be difficult to manage. For example, information on the timing and intensity of recent fires will indicate whether particular aspects of the fire regime are occurring less or more frequently than desired.

There are three particularly important caveats for this approach. Firstly, the hypothesised relationship between variability in fire regimes and biodiversity (Keeley 1991, Gill and McCarthy 1998) has rarely been evaluated (e.g. Jackson 1968, Keith and Bradstock 1994, Morrison et al. 1995). The influence of different aspects of 'pyrodiversity' on biodiversity, and the spatial and temporal scales of that influence, are not well known (Clarke 2008). Secondly, a uniform distribution of successional stages was used as the goal for illustrative purposes, but this is not necessarily a desirable configuration. Different vegetation types are likely to have different optimal age class distributions (Clarke 2008). A greater propensity of early-successional stages may impart greater benefits for fire suppression, while the influence on biodiversity will depend on the relative abundance of species in different age classes. The section below (*Geometric mean abundance*) describes how the optimal distribution of successional stages might be determined if the relative abundance of species in each were known. This shows that the optimal allocation among stages might be far from uniform where the aim is to maximise biodiversity.

A third important caveat is that the vegetation dynamics have simply been described in terms of the time since the last fire. The disturbance history prior to the last fire can also be influential, along with the intensity of the disturbances. Sufficiently low-intensity disturbances can leave important legacies on sites, such as hollow-bearing trees and large logs, that can have important influences on biodiversity (Franklin et al. 2000). The model of Richards et al. (1999) can be modified to account for different descriptions of the structure of the vegetation (e.g. combinations of times since the most recent and second-most-recent fires, and the intensities of those).

Measures of resilience based on remote sensing

Remote sensing data have wide spatial coverage, and relatively consistent collection methods, so they can potentially be useful for monitoring. For example, Washington-Allen et al. (2008) measured NDVI (Normalised Difference Vegetation Index, a vegetation index based on the difference in the reflectance of incident near infrared and red radiation detected by satellites) to monitor changes in vegetation in response to drought. The mean and variance of the NDVI measures both the average condition of the vegetation and also the spatial variability of the vegetation. Large variation suggests that some parts of the landscape may have reduced vegetation cover. Different aspects of the vegetation can be measured, including the rate of change at points in the landscape, and the degree to which the vegetation differs from a reference condition. The degree to which different vegetation types return to a reference condition, and the rate of that change, can be measured. While remote sensing imagery measures previous changes in vegetation, it cannot necessarily forecast future changes, and changes below canopies of trees are likely to be obscured.

Measures of resilience based on modelling

Resilience has a diversity of dimensions. Further, the aspects of a system that need to be resilient will vary from place to place. In these circumstances, and when *a priori* measures of resilience are unavailable, it is not surprising that universal measures of resilience have not been established. In the absence of empirical studies, or even where some empirical data exist, *a priori* prediction of resilience requires the use of models. In this section, three different indicators that might be useful for measuring aspects of ecological resilience are described.

1. Divergence from the goal age structure

In the example of managing fires to maintain a diversity of age structures, an implicit assumption is that a diversity of age classes makes the system more resilient to change. This is a simple model (age class diversity begets resilience), but suggests one possible way of measuring resilience. The divergence between the observed age structure $f(x)$, and the goal age structure $p(x)$ can be measured by the relative entropy of the age class distributions (Kullback and Leibler 1951):

$$D = \sum_{x=1}^{n} f(x)\ln(f(x)/p(x)).$$

For example, the value of D for Foothills Forest is:

$$\begin{aligned} D_{\text{Foothills}} &= 0.015 \times \ln(0.015/0.33) + 0.035 \times \ln(0.035/0.33) + 0.95 \times \ln(0.95/0.33) \\ &= 0.87. \end{aligned}$$

Compared with the other EFGs (Table 2), the age class of the Foothills Forest diverges by the greatest amount from its goal age structure (in this case one-third of each age class), and its relative entropy score is the largest. EFGs that are closer to their goal age structure have lower relative entropy scores. Thus, the relative entropy of each EFG provides one possible measure of the community's resilience. Increases in the relative entropy would indicate increasing divergence from the goal age structure. This would indicate to managers whether management is on target to reach the goal age structure.

The diversity of successional stages provides a degree of latitude (Figure 1) in response to disturbance. If the entire landscape consisted of a single successional stage, an unplanned event that adversely affected that particular age class would cause large changes. A diversity of successional stages provides some insurance in the face of change.

Table 2. The current age structure of the major EFGs of the Otways study area, and the relative entropy of each, which measures the divergence between the current age structure and the goal age structure. In this case, the goal age structure was assumed to be one-third of each successional stage.

Ecological Fire Group	Successional stage			Relative entropy
	Early	Mid	Late	
Damp Scrub	0.08	0.49	0.43	0.19
Foothills Forest	0.01	0.03	0.95	0.87
Forby Forest	0.13	0.27	0.60	0.18
Heathland (sands) – Default	0.14	0.43	0.43	0.10
Moist Forest	0.04	0.28	0.67	0.34
Tall Mist Forest	0.45	0.55	0.00	0.41
Tall Mixed Forest (Eastern)	0.06	0.12	0.82	0.52

2. Susceptibility to short fire intervals

A second possible index of resilience is the susceptibility of each community to fire intervals that are too short. This can be measured by the area of a community that is expected to be burnt each year by a fire with too short an interval. Assume we can divide the area of each EFG into equally sized cells (e.g.. grid cells), and focus on the n cells for which the time since fire is less than the minimum tolerable fire interval (TFI). Let the annual probability of (unplanned) fire for cell i equal p_i. Then the expected annual area burnt by fires with a fire interval less than the minimum TFI will equal the sum of annual fire probabilities over these n cells: $S = \sum_{i=1}^{n} p_i$. This can be expressed as $S = pn$, where p is the probability of fire averaged over the n cells. Thus, this measure of susceptibility increases (resilience decreases) with the average annual probability of unplanned fire p, and the area that is less than the minimum tolerable fire interval n. Prescribed fires in particular parts of the landscape might be able to reduce p, but will tend to increase n, meaning that there is a trade-off between using prescribed fires to increase the resilience of ecosystems. Prescribed fires will increase this aspect of ecosystem resilience (decrease S) provided that the increase in n is offset by a sufficiently large decrease in p. The variable S can measure the susceptibility imparted by different fire management scenarios, and it can be used to monitor changes in susceptibility of landscapes over time. While p_i (and hence p) can be estimated from previous unplanned fire events (e.g. Figure 4), modelling (e.g. using Phoenix, Tolhurst et al. 2008) would be required to determine how p_i is influenced by fire management (e.g. the location, size and proximity of prescribed burns) under particular weather conditions.

Susceptibility to short fire intervals provides one measure of resistance (Figure 1) to environmental change. If large areas of a landscape are younger than the minimum tolerable fire interval, and if the probability of fire in these areas is large, fires would be expected to have large negative influences on biodiversity.

3. Geometric mean abundance

An indicator of biodiversity

A third measure of resilience in the context of fire management can measure how the relative abundance of species changes. As previously discussed, resilience might change as the relative abundance of species changes, even when the number and identity of species in a community is constant (e.g. Wittebolle et al. 2009). A parallel focus of research has developed indices that can measure changes in biodiversity, particularly as a reporting mechanism for the Convention for Biological Diversity (Buckland et al. 2005). The geometric mean abundance, or a function of it, tends to be used most frequently (e.g. Loh et al. 2005). The geometric mean has a number of useful properties (Buckland et al. 2005). It is sensitive to changes in the relative abundance of sets of species; a decrease in the abundance of a rare species by N individuals must be compensated by more than an increase of N individuals of a common species. The geometric mean is sensitive to proportional changes in abundance; a 20% decline of one species (a multiplicative change of 4/5) must be compensated by a 25% increase in a second (a multiplicative change of 5/4; the reciprocal of the multiplicative change of the declining species). Additionally, relative changes in abundance are sufficient to calculate an index if the proportion of the population being sighted is constant over time. While the proportion of the population being sighted must remain constant over time (or be accounted for if it does change), this proportion can vary among species. This is particularly useful because often estimates of changes in abundance will simply provide relative measures rather than actual population sizes (e.g. MacHunter et al. 2009).

In addition to having useful heuristic properties, the geometric mean can also be related to the viability of species based on theory. In habitat that is unsuitable for a particular species (e.g. caused by inappropriate fire regimes), a deterministic decline at a relatively constant rate might be reasonably expected. In the presence of deterministic population decline, the mean time to extinction of a population is proportional to the logarithm of population size. Thus, the arithmetic mean of the logarithm of abundance $\ln(N_i)$ of n species within a community will be proportional to the average time to extinction of populations that face deterministic extinction:

$$\overline{T} = \frac{k}{n}\sum_{i=1}^{n}\ln(N_i),$$

where k is a constant of proportionality.

Note that the geometric mean abundance is given by:

$\overline{N}_{geom} = \exp(\frac{1}{n}\sum_{i=1}^{n}\ln(N_i))$. Thus, the logarithm of the geometric mean abundance is $\ln(\overline{N}_{geom}) = \frac{1}{n}\sum_{i=1}^{n}\ln(N_i)$. Substituting this expression into the equation for the mean time to extinction shows that the mean time to extinction of species in the community in the presence of deterministic extinction ($\overline{T}$) is expected to be linearly related to the logarithm of the geometric mean abundance ($\overline{T} = k\ln(\overline{N}_{geom})$). Thus, the geometric mean abundance of species has i) theoretical support as a predictor of species extinction (there is also some empirical support for this based on observed local extinctions (McCarthy et al. in prep.), ii) useful practical properties as an indicator of biodiversity (Buckland et al. 2005), and iii) been used previously as an index of biodiversity (Loh et al. 2005).

Defining the desired age structure

The previous discussion illustrates that the geometric mean abundance of species might provide a useful indicator of the viability of species in a community. If this is the case, a second role for the geometric mean abundance might be to help define the desired age structure of different EFGs. For example, consider the case of three focal species for which the relative abundances in different successional stages is given as in Table 3. The species A, B and C represent late-, early- and mid-successional species because their abundance is greatest within these stages. In this case, the relative abundance of species *i* within a landscape, averaging over all successional stages, will depend on the proportion of the landscape area that is early- (a_E), mid- (a_M) or late-successional (a_L); this abundance is given by $R_i = r_{E,i}a_E + r_{M,i}a_M + r_{L,i}a_L$. Note that $a_E + a_M + a_L = 1$. We can find the optimal allocation of a_E, a_M and a_L that maximises the geometric mean of R_i averaging across the species in the community. In the case of Table 3, the optimal distribution of successional stages is $a_E = 0.34$, $a_M = 0.35$, and $a_L = 0.31$, close to one-third in each stage. In this case, there is slightly increased weighting to the mid-successional age class because this supports at least moderate abundances of all species. There is slightly less weighting to the late-successional stage because this age class has low abundance for species B and C, while the early-successional stage has moderate abundance for species C. However, if there is a second late-successional species in the community ($r_{E,D} = 1$, $r_{M,D} = 1$, $r_{L,D} = 10$), the optimal allocation of land to successional stages changes, being $a_E = 0.28$, $a_M = 0.12$, and $a_L = 0.59$ in this case. The extra late-successional species means the late-successional stage is weighted more heavily. The age structure that maximises the geometric mean abundance varies depending on the number of species that prefer particular successional stages and the relative abundance of those species in the different successional stages.

Table 3. A hypothetical example of the relative abundance of species in the different successional stages. The mix of abundances influences the allocation of age structures that maximises the geometric mean abundance of the different species.

Species (i)	Relative abundance in successional stages Early ($r_{E,i}$)	Mid ($r_{M,i}$)	Late ($r_{L,i}$)
A	0	3	10
B	10	2	1
C	4	10	2

As an example application of determining the optimal allocation of land to successional stages, I assigned relative abundances for the 32 vertebrate fauna key fire-response species of Foothills Forest (Table 4) based on the assumed fire-response curves provided by MacHunter et al. (2009). Note that this is used for illustrative purposes. In a particular area, the fire response may not match that displayed in MacHunter et al. (2009), and my assignment of relative abundances to age classes was based on a subjective assessment of the diagrams in MacHunter et al. (2009). These assignments should be checked against data for each forest type in the relevant region, and monitored in future to evaluate the assumed relationships. With these caveats in mind, most of the key fire-response species in this case study are most abundant in the late-successional stage. Further, the species that are most abundant in the early- and mid-successional stages are reasonably abundant in the late-successional stage, with the red-browed finch being one exception (an A1 response species). In this case, the allocation that maximises the geometric mean abundance of these species is $a_E = 0.0$, $a_M = 0.04$, and $a_L = 0.96$. The geometric mean relative abundance of the 32 species in this case is 7.1. The heavy weighting to the late-successional stage reflects the large number of species with high abundance in this stage. Under a uniform allocation across stages ($a_E = 0.33$, $a_M = 0.33$, and $a_L = 0.33$), the geometric mean relative abundance is reduced to 5.6, reflecting decreased viability (decreased resilience). A greater propensity of the early stage ($a_E = 0.5$, $a_M = 0.25$, and $a_L = 0.25$) would further reduce the geometric mean abundance to 4.8. Such influences on the mean abundance of species could be traded-off against the other benefits of changing the age structure (e.g. the fire-protection benefits of having more recently burnt areas in the landscape).

Table 4. Relative abundance of vertebrate fauna key fire-response species versus successional stage and the number of species in Foothills Forest, derived from the assumed relationships in MacHunter et al. (2009). The fire-response code describes changes in abundance with time since fire, corresponding to the relationships in MacHunter et al. (2009).

Fire-response code	Number of species	Relative abundance in successional stages Early ($r_{E,i}$)	Mid ($r_{M,i}$)	Late ($r_{L,i}$)
A1	1	10	2	1
A2	6	10	7	5
B1	1	1	8	7
B2	6	1	10	5
B3	18	1	5	10

The geometric mean abundance of species provides a useful indicator of the state of biodiversity, so it could be a focus of monitoring. In particular, for species that respond to time since fire, monitoring could assess the assumed relationships in MacHunter et al. (2009) between time since fire and the relative abundance of species. If these relationships are supported, then the relative abundance measures can be extrapolated across the landscape, and the geometric mean abundance of these indicator species reported. Declines in these key fire-response species would suggest decreasing resilience in response to fire management.

The geometric mean abundance of species indicates how the biodiversity is changing over time, providing a relative measure of precariousness (Figure 1). As the abundance of species declines, they become more susceptible to environmental change, and the risk of unacceptable declines increases.

In the presence of climate change, managers may be operating against a shifting benchmark in which risk of fire, the response of biodiversity and the effectiveness of management are changing. Decision frameworks such as those developed by Richard et al. (1999) can be modified to account for expected changes in probabilities of fire with time by including them explicitly in the analysis. However, uncertain or unexpected changes are more difficult to address.

There is uncertainty in the assumed relationships between biodiversity and time since fire (or other aspects of the fire regime such as intensity). While monitoring and evaluation of these assumed relationships is vital, accounting for this uncertainty in current management decisions is important. This appears to be an open area of research – how to define fire management objectives for biodiversity when the relationships between fire and biodiversity are uncertain? Assuming particular relationships (e.g. abundance versus time since fire, as done in this report) may be risky. Species may have unexpected declines if they do not respond in the assumed way. Similar uncertainty exists in other aspects of the fire management problem, such as when determining the asset protection benefits of fuel-reduction burning. Further research on how to incorporate uncertainty, and the influence of this uncertainty on management decision-making, seems warranted.

Resilience, adaptive management and uncertainty

Adaptive management is a structured, iterative process of decision-making in the face of uncertainty (Holling 1978, Walters 1986). It aims to reduce uncertainty by monitoring outcomes of management actions and inaction. In this way, decision-making aims to achieve particular management objectives while gaining information that improves future management. Adaptive management is often characterised as a continual cycle of 'learning by doing' (Figure 9).

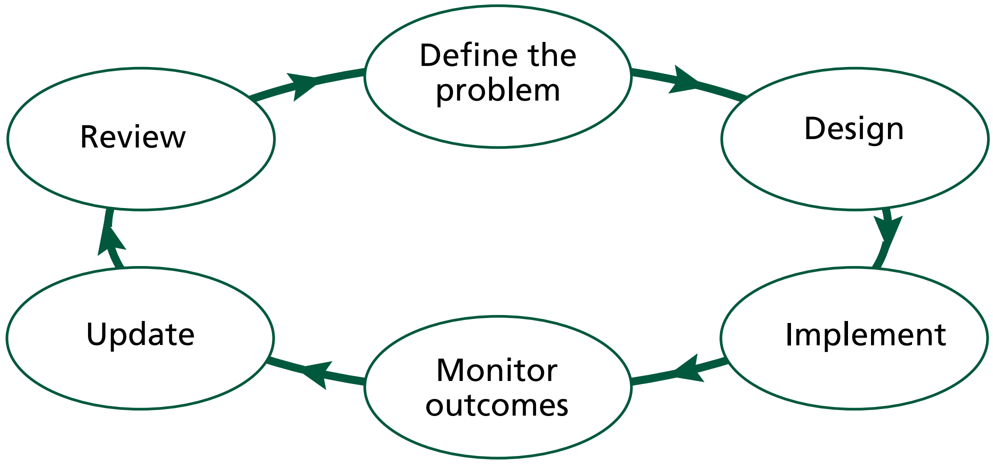

Fig. 9. A six-step adaptive management cycle in which managers and stakeholders i) define the problem and its uncertainties, ii) design management and monitoring strategies based on predicted responses to management actions and inactions, iii) implement the management strategies, iv) monitor the outcomes of management actions and inactions, v) update knowledge about the effects of different management strategies, and vi) review and re-evaluate the management questions in the light of the updated knowledge (modified from Whelan 2004, Cawson and Muir 2008).

Active adaptive management places an explicit value on learning about the effectiveness of management. Management actions might be modified and the outcomes monitored, with the specific goal of improving knowledge that can contribute to better management in the future. It contrasts with passive adaptive management in which monitoring of management effectiveness still occurs, but management is not modified with the aim of increasing learning. Instead, learning occurs serendipitously, depending on the suite of actions being taken and the environment in which they are implemented. Any knowledge is then incorporated into management plans (Parma et al. 1998, Shea et al. 1998, 2002). There are a range of examples of passive and active adaptive management in the ecological literature (e.g. Johnson et al. 1993, Varley and Boyce 2006, McCarthy and Possingham 2007, Nichols et al. 2007), although it is rarely applied in practice, largely because targeted monitoring of management effectiveness is rare. Further, active adaptive management presents major conceptual and theoretical challenges. Active adaptive management involves designing conservation measures in such a way that managers can learn efficiently about the system for which they are responsible so that future management is improved, bearing in mind the needs of managing the system in the present. It is recognised that experimentation is important in environmental management (Ferraro and Pattanayak 2006), but it is not clear how conservation resources should be split between learning through experimentation and management based on what is known currently. Mathematical analysis of adaptive management problems can help resolve these questions (e.g. McCarthy and Possingham 2007), but optimising active adaptive management programs for all but the simplest problems is computationally difficult.

One of the key insights of research on adaptive management is that the value of learning through experimentation can be expressed in terms of the expected benefit to the management objective (Walters 1986). It is possible, at least in theory, to assess how resources should be allocated between learning about the effectiveness of management and actually managing the system. The concept of valuing learning in terms of improved management is well understood (Walters 1986), although it is mathematically difficult to assess the trade-off between allocating resources to conduct well-designed scientific studies and using resources that maximise the expected conservation outcome.

Some of the greatest uncertainties about resilience of vegetation communities to changed fire regimes to relate the degree by which fire regimes can change and still provide resilient vegetation communities. If the active adaptive management strategy is different from the passive strategy, it is likely to aim to push the system into a non-resilient state to learn about the location of this tipping point and the ability to recover (Hauser and Possingham 2008, Moore et al. 2008). However, this is unlikely to be a desirable outcome, and it would not necessarily show how far the system could change but still remain resilient. Further, and perhaps more importantly (given that the cost of entering non-resilient states can be accommodated in the analysis), determining the optimal active adaptive management strategy may be computationally difficult for realistically-sized problems that contain many uncertainties. In these circumstances, passive adaptive management may be preferable, and still provide outcomes that are similar to those based on active adaptive management (McCarthy and Possingham 2007).

While optimising adaptive management might be computationally challenging, adaptive management ideas can still be applied without conducting these optimisations. In particular, adaptive management can help to focus attention on important research questions that are likely to be answered through formal monitoring programs. Both passive and active adaptive management rely on developing alternative models of how the system being managed might work. These models can represent different opinions of the important functional relationships between elements in the system (e.g. the influence of predation, competition, rainfall and fire on the dynamics of different species) and uncertainty about the strength of those relationships (e.g. parameter uncertainty). As part of adaptive management, monitoring the outcomes of management action and inaction and evaluating the data helps to refine parameter estimates and discriminate between the competing models. This improved understanding then contributes to an iterative process of decision-making and monitoring.

Adaptive management does not necessarily remove the risk of management failures. Failures can occur if the range of models being considered is not sufficiently broad (Peterson et al. 2003), so a premium is placed on diversity of opinion and critical analysis of the alternatives. However, even when using a broad range of models, unplanned events can lead to undesirable management outcomes. Adaptive management simply aims to reduce the occurrence of undesirable outcomes now, and into the future. Perhaps one of the biggest advantages of adaptive management is that it provides a framework for thinking rigorously and in a focussed manner about how the system works, the benefits of management and what needs to be monitored.

Whelan (2004) identifies numerous examples of fire management questions for which the answers are generally imprecise (see also Cawson and Muir 2008). These questions concern the effectiveness of different fire regimes (e.g. intervals, season, intensity) for reducing risks of unplanned fire, and the influence of different aspects of fire regimes on different aspects of biodiversity. Managers with an imperative to both protect human life and property, and also manage biodiversity do not have the luxury to wait until these questions are answered before making decisions about how to allocate resources to fire management. Therefore, adaptive management provides a mechanism for managing now in the face of uncertainty and planning to reduce the uncertainty through well-designed monitoring programs (Whelan 2004, Cawson and Muir 2008). A key element of adaptive management is that monitoring focusses not simply on areas where there is large uncertainty, but on those uncertainties that have the biggest influence on management. These areas of uncertainty might not be the most interesting from a scientific perspective. Further, the optimal management and monitoring strategy might not have the maximum possible statistical power because the experimental management provides management outcomes, not just knowledge (McCarthy and Possingham 2007). However, these monitoring programs must have sufficient rigour that they can answer the questions for which they were designed (Walters 1997). It is usually important to monitor changes not just in response to management actions, but also in places where management has not been undertaken. These sites provide a point of contrast with the influence of management actions and are fundamental to standard experimental design. Such sites at which management actions are not occurring are particularly important in the presence of other factors, such as climate change, that are largely beyond the control of management.

Cawson and Muir's (2008) review of adaptive management is a good overview, but some additional points are worth noting. While Cawson and Muir (2008) note the importance of monitoring to reduce uncertainty, monitoring should also aim to address topics that influence management and where the monitoring is likely to help resolve the management uncertainty. In some cases, the statistical power will not be sufficient to remove the uncertainty. In other cases, management may have little ability to influence the outcome. In these cases, monitoring as part of adaptive management will not be beneficial. For example, consider the role of fire season on biodiversity. While there is considerable uncertainty about how the season of occurrence may influence biodiversity in a particular area, in some ecosystems there might be little scope for managers to control when fires occur. Adaptive management of fire season might be useful in those ecosystems where the season of planned burning can be controlled (e.g. spring versus autumn), but less so where fires are largely restricted to a single season. For example, burns might be largely restricted to autumn in some ecosystems when fuel is mostly too wet in spring. Similarly, managers only have limited scope to control intensity of fires, so information on responses to fires of different intensities might be difficult to incorporate into deciding on appropriate management strategies. Adaptive management needs to consider these issues before decisions are made about appropriate monitoring strategies.

In the context of managing fire regimes, one uncertainty is how the distribution of successional stages of vegetation through the use of prescribed fires influences biodiversity and other assets (e.g. water yield) directly. Further, changing the spatial arrangement of prescribed fires might have direct influences on biodiversity, and possibly indirect influences by changing the probability and consequences of unplanned fires. While these influences can be modelled (e.g. using predictions from Phoenix, Tolhurst et al. 2008) and combined with information on how water yield and biodiversity respond to fires, the predictions are invariably uncertain (e.g. Clarke 2008, MacHunter et al. 2009). Adaptive management would seek to establish areas with different fire management histories and to monitor the response to this fire management. Learning about some responses, such as the impacts of high-intensity unplanned fires, will be slow given the infrequent occurrence of fires of this type and the inability to control where these events occur. However, learning about other responses might occur faster, such as how biodiversity responds to time since fire or different spatial arrangements of successional stages. These latter uncertainties and other important research questions might be resolved by further monitoring with sufficient spatial and temporal resolution (Clarke 2008). In all cases, learning will be improved where different management actions and inactions are compared.

Conclusions and recommendations

Resilience is a relatively vague concept because it lacks a general operational definition that permits measurement. Systems that are non-resilient can often be identified after the fact, but *a priori* prediction of whether a particular system is resilient to particular management strategies is difficult. In this case, models are required to predict resilience *a priori*. The theory of resilience suggests that diversity of various types, including functional diversity and redundancy of species, is likely to confer resilience to ecosystems. Only a small number of empirical studies have tested aspects of these ideas. Maintaining diversity of landscape age structure that arises from fire management and promoting diversity of fire regimes is hypothesised to contribute to ecosystem resilience. The susceptibility of landscapes to fires with too short an interval can be predicted, as can the geometric mean abundance of species. These three metrics (the diversity of landscape age structure, the susceptibility to disturbance and the geometric mean abundance of species) are hypothesised to be useful indicators of ecosystem resilience, reflecting the aspects of resilience referred to as latitude, resistance and precariousness (Figure 1). Monitoring could focus on these indicators, both to track environmental change and also to evaluate their use as indicators of resilience.

Quantitative tools are available to help managers make decisions that promote diversity. One of these, developed by Richards et al. (1999), was demonstrated for fire management in the Otways. However, the response of biodiversity to particular fire regimes, the ability of different fire regimes to influence the incidence and effects of unplanned fires, and the resilience of systems to different fire regimes, are poorly understood. In the face of this uncertainty, managers must still develop and implement fire management plans. However, a rigorous system of monitoring that is directed to the most important uncertainties can help improve management. This process of learning by doing is termed ‘adaptive management’. It provides a mechanism by which managers can invest in monitoring that will pay for itself in terms of improved management. Monitoring should focus on those aspects that are uncertain, where the uncertainty clouds the decisions, and where the monitoring is expected to help reduce the uncertainty sufficiently to improve the management decision.

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