Felderhof, Leasie (2007) ***The fire patchiness paradigm: a case study in northwest Queensland.*** **PhD Thesis, James Cook University.**

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https://doi.org/10.25903/cg1t%2Dj980

The Fire Patchiness Paradigm:

A Case Study in Northwest Queensland

Thesis submitted by

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in April 2007

for the degree of

Doctor of Philosophy

in Environmental Science

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Leasie Felderhof

STATEMENT OF CONTRIBUTION OF OTHERS

Ergon Energy

- Operating Expenses ($80,000).

Tropical Savannas Management CRC:

- Post-Graduate Research Scholarship.

James Cook University:

- Library access,
- General student support and supervision,
- Conference attendance.

CSIRO (Atherton)

- Library access and use of drying ovens.

QPWS

- Office space in early stages of the project,
- Accommodation at Lawn Hill National Park,
- Access to LANDSAT imagery used in Chapter 2,
- Library resources,
- Use of herbarium (Mareeba).

ACKNOWLEDGEMENTS

It is not conventional to thank family members first, but this thesis would not have been commenced or completed without the full support of my husband Bill, and the tolerance of my children, Mariska, Will and Luke. Producing this work involved tenacious navigation through cancer treatments, broken bones, heart operations, extended family and cyclones. Their positive spirits and mutual support were invaluable.

The expert advice and guidance provided by my supervisors Prof. David Gillieson (James Cook University) and Drs. Garry Cook (CSIRO), John Ludwig (CSIRO), and Jeremy Russell-Smith (Tropical Savannas Management CRC) was very much appreciated. I thank them for their friendship and feedback.

This thesis involved extensive field work in remote locations, often under arduous conditions. A long list of people provided cheerful assistance, making it more fun than work. Thanks to: Mike Ahmet, Sarah Atkinson, Ros Blanche, Christine Crafter, Graham Harrington, Tony Johnstone, John Ludwig, Bev Masasso, Mike Mathieson, Doug McConnell, John Prince, Geoff Smith, Mark van Ryt, Brian Venables, Amy Wentz and Paul Williams. Kirsten Almer, Sarah Atkinson, Catherine de Chermont, Rebecca Felderhof and Gail Lowe assisted with data processing for various chapters and John Ludwig provided advice on statistical analysis. Les Searle is thanked for ordering satellite imagery and getting me started with ERMapper®. The comments and expert opinion provided by Dr. Gabriel Crowley, Dr. John Woinarski, Dr. Graham Harrington, Dr. Geoff Smith, Dr. Mike Mathieson, Dr. Chris Clague, Dr. John Winter and Keith MacDonald for the fauna fire-response database (Chapter 5) helped to progress my thinking and I am grateful for the time they allocated to this project. Discussions on fire management with Dr. Stephen Garnett and Peter Thompson in the early stages were also stimulating. Graeme Chapman is also thanked for generously allowing me to use his Kalkadoon

Grasswren pictures on the title page and in related presentations. The final collation of the thesis greatly benefited from formatting and editorial assistance provided by Tina Lawson and Annette Bryan.

Ergon Energy, the Tropical Savannas Management CRC and James Cook University generously provided funding for fieldwork, computing resources and conference attendance. Queensland Parks and Wildlife Service gave permission for some of the research to be undertaken in conjunction with my normal work duties, which was appreciated. Further, the fieldwork would not have been possible without the co-operation of landholders in northwest Queensland. They willingly provided access to their properties and information on local fire management practices, for which they are thanked.

Final thanks go to my cycling friends, who are not the least bit interested in fire ecology. Apparently there is more to life…. They are thanked for letting me discuss ecological theories with them anyway, and for keeping me on the bike when the going got tough.

DEDICATION

This thesis is dedicated to the memory of Jill Landsberg. Jill drew my attention to the project and casually asked, "You wouldn't be interested, would you?". This single remark had an unprecedented effect – and resulted in the production of this thesis.

ABSTRACT

The Fire Patchiness Paradigm

Research into fire ecology has culminated in 'the patchiness paradigm'. This is the view that numerous small fires, with variety in fire timing, frequency and intensity, will lead to habitat diversity across the landscape, thereby benefiting species conservation. The paradigm recognises that fire is instrumental in shaping the Australian environment, and that no single fire regime will suit all species. As a result, land holders wanting to adopt ecologically sustainable fire management practices are advised to develop and maintain fine-grained landscape patchiness using fire. However, there is no guidance on optimal fire size, level of internal fire patchiness or desirable fire frequency. Having ill-defined objectives is incompatible with 'adaptive management', the approach espoused for managing in the face of inadequate information and uncertainty. Adaptive management, or learning by doing, requires clear management objectives and careful monitoring. When, how often, and how much should manager's burn, and how should they evaluate success? Although adaptive management accommodates uncertainty, the scale and internal patchiness of proposed fire operations need to be articulated in order to commence the cycle with a 'best guess' management target. Addressing this issue and bridging the gap between ecological understanding and applied land management is the critical next step in fire ecology. This thesis focuses on this concern.

The case study area was the Mount Isa Inlier, a semi-arid bioregion in northwest Queensland, Australia. The study concentrated on the fire ecology of spinifex/snappy gum woodlands, the dominant vegetation type in the bioregion. Remote sensing technology was used to examine landscape fire patterns and to identify key drivers of these patterns. Field surveys were undertaken to determine

regenerative responses of the vegetation and to investigate spatial variability between and within fires. To investigate the potential effects on fauna, a fauna fire-response database was compiled using expert opinion. Species were then classified according to their fire sensitivity based on species' refugium requirements during a fire and vegetation maturation stage requirements after a fire. The predicted response of fauna was tested in the field using birds as a sub-group. The results were interpreted in terms of setting objectives for adaptive management, with recommendations on a target fire size and associated monitoring.

The Mount Isa Inlier was found to have greater affinity with arid Australian landscapes than mesic savannas. Fire scars mapped over a six-year period (1998-2003) showed that the total area burnt per year was related to the strength of the preceding wet season. Post-fire changes in the vegetation were highly predictable. There was a flush of annual and ephemeral species after the first rain. The abundance of these species decreased over time, while the cover of perennial species increased. There was insufficient fuel for fire to spread for at least three years from the previous fire. By then, most ephemeral species have set seed, and perennial species have started to produce seeds. Resilience to fire was enhanced by the inherent variability within burnt areas. Vegetation at early recovery stages co-existed with patches of vegetation at later recovery stages. Mature plants in unburnt patches provide reproductive material that ensures individual species remain in the system. Collectively, these data provided an understanding of fire-mediated vegetation dynamics in spinifex/snappy gum communities.

When the fire-sensitivity of fauna species was tested, birds relying on long-unburnt spinifex during and after fire were more vulnerable than generalist species. Additional field studies were undertaken to investigate the internal fire-patchiness requirements of fire-sensitive species. The ideal characteristics for unburnt patches within fire scars could not be defined. Nonetheless the approach provided sufficient insight for a target fire size to be proposed. Restricting fire size to around 100 ha was considered acceptable to conserve birds in the bioregion. It is recommended that a suite of fire-sensitive species be used for ecological monitoring, based on the objective process used in this study.

Fine-scale satellite imagery (IKONOS, 1-m^2 – 4-m^2 pixel resolution) was investigated as a tool for quantifying fire patchiness. If fuel load or fire severity could be mapped, this would provide the initial step. Mapping fuel loads at this scale would also give accurate and spatially explicit fire histories for different areas. This would benefit researchers interested in characterising fire regimes as it would allow for successive monitoring. Field data on fuel load and fire severity were compared to reflectance values recorded by the IKONOS satellite. A fuel load map was produced, but the inherent uncertainty in remote sensing processes meant that results were generalised to areas with high, medium or low fuel load. Although this can assist managers to identify areas of high fire hazard, its day-to-day use for monitoring fire patchiness is questionable. Mapping fire severity was not successful and remains the key fire regime variable not yet captured by remote sensing. The fine-scale mapping components of this study supported the notion that restricting fire size should be emphasised as a management goal in northwest Queensland. Internal fire-patchiness was inherent and difficult to specifically plan for. Fire size can be readily monitored using existing remote sensing techniques.

The major contribution of this work is that it provides a way to mesh the patchiness paradigm with practical land management. By tailoring fire size to the needs of the most fire sensitive species, the remainder are likely to be catered for, by default. Fire-sensitive species can be identified by classifying species according to their requirements during, and immediately after, fire. The method tested here proved to be useful and can be applied in different environments and at different scales.

This is one of the few studies that endeavours to quantify the level of patchiness to which managers might aspire. It demonstrates a strategic approach that integrates well with adaptive management. Thus, the findings provide a way to progress from setting management goals based on ecological principles, such as ‘a fine scale mosaic’, to setting more specific targets based on ecological understanding and a coherent process. Further, the thesis provides important information on the fire ecology of spinifex/snappy gum woodlands in the Mount Isa Inlier. Fire-related research is scant for this area, so this information provides a starting point for ecological fire management in the region.

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1. INTRODUCTION

Fire, herbivores and climate are major determinants of terrestrial vegetation (Bond 2005). These elements have different degrees of influence in different ecosystems. In the savannas of northern Australia, fire dominates – it depresses biomass from reaching its potential based on climate alone, and fire-effects flow on to other ecological processes (Andersen *et al.* 2003a). Managing fire is fundamental to managing savanna landscapes.

This thesis investigates issues related to fire ecology and fire management in northwest Queensland, Australia. It focuses on the Mount Isa Inlier, a bioregion where information related to fire ecology is scant. Virtually no fire-related research has been undertaken in the area, or for the semi-arid environment in which it occurs. This poses problems for land managers seeking to adopt ecologically sustainable fire management practices. What should they aim for? When should they burn? How should they gauge success? This thesis utilises a typical planning cycle as a framework for research; it provides information for both planning fire management and monitoring results. The theoretical emphasis, however, is on the 'patchiness paradigm'. This is the idea that numerous small fires, with variety in fire timing, frequency and intensity, leads directly to habitat diversity, which benefits species conservation. Building on this premise, fire managers aim to implement fire regimes that develop and maintain fine-grained landscape patchiness. But ecologists come to an impasse when recommending appropriate patch sizes for species conservation. Details on the desirable scale of patchiness are sketchy, preventing land managers from setting measurable fire management goals. The thesis further explores a methodology for determining an appropriate level of fire-mediated landscape patchiness and investigates issues related to its subsequent monitoring using remote sensing.

Landscape fire patterns within the Mount Isa Inlier were examined using remote sensing, together with field surveys, to determine regenerative responses in the vegetation. A database on the likely responses of fauna to fire was compiled through expert opinion and then used to explore options for setting ecological fire management objectives. Birds were selected as a group to test the predicted response of fauna in the field, and surveys were undertaken on burnt areas, unburnt areas and on smaller unburnt patches within burnt areas. Birds were used because they are responsive to land management practices (Landsberg *et al.* 1997; MacNally *et al.* 2004) and have been recommended as indicators for environmental monitoring at landscape and bioregional scales (Smyth & James 2004). These data contribute to our broader understanding of fire ecology in northwest Queensland and provide essential information for ecological fire management in this area.

1.1 Ecological Fire Management

Ecological fire management is the active use of fire in nominated areas to achieve specified ecological objectives. The concept has arisen from many decades of research into fire ecology, which has led to acknowledgement that fire is instrumental in shaping the Australian environment (Bradstock *et al.* 2002; Whelan 1995; Whelan *et al.* 2002; Woinarski 1999). The accumulated knowledge has resulted in a change in opinion and actions by European settlers in Australia over the past 200 years. Attitudes have moved from fire as a force to be feared and prevented to a recognition that fire is a natural ecological process and is required for maintaining biological diversity (Bowman 2003; Russell-Smith 2002). Appropriate fire management is now considered critical for species conservation (Bradstock *et al.* 2002; Gill *et al.* 1981; Gill *et al.* 1999) as changed burning practices since European settlement are implicated in vegetation change (Bowman & Latz 1993; Bowman & Panton 1993; Russell-Smith *et al.* 2000; Russell-Smith *et al.* 2002;

Russell-Smith *et al.* 1998) and species loss (Franklin 1999) across broad geographic areas. Aboriginal burning tended to be highly patchy and provided for the development of relatively fine-grained habitat (Russell-Smith 2002). In contrast, the contemporary pattern in the tropical savannas is for areas to be burnt frequently by extensive, intense fires or seldom burnt. Thus, Russell-Smith (2002) suggested that “by far the most important requirement for conservation of biodiversity is the imposition of patchy management regimes”. The recognised link between fire management and species conservation has led some land management agencies and land managers to review their fire management practices. Where historic emphasis has been on active fire management to protect life and property (by reducing fuels and therefore fire hazard), managers concerned with conservation now have broader fire management objectives – to protect life and property as well as to maintain ecosystem function for species conservation (ESA; FEWG 1999; Melzer & Clarke 2003; Rose *et al.* 1999). This thesis is concerned with the latter, that is, the active use of fire to alter plant and animal community composition and habitat structure to achieve species conservation outcomes.

The need for active fire management has been recognised for some time (Gill *et al.* 1981); however, uptake of ecological fire management as a tool has been slow. This can be ascribed to three issues: (1) The legal and social requirements to protect life and property remain paramount; (2) The limited resources available for conservation management; and (3) The lack of knowledge on fire/species interactions, which are extremely complex. Different species respond to fire in different ways, and individual fires differ depending on local environmental conditions. Further, the effect of one fire can affect the nature of subsequent fires, so it is difficult to set specific management objectives. The fire regime requirements of most species are not well known, if at all.

Where research has identified requirements for rare or threatened species, ecological fire management has moved past the conceptual stage. Examples include targeted management for threatened species such as the Golden-shouldered Parrot (Garnett & Crowley 2003), Mallee Fowl (Benshemesh 2000), Eastern Bristlebird (Holmes 1998), Ground Parrot (McFarland 1991), Partridge Pigeon (Woinarski 2004), and others. However, where fire has been implicated in causing a general decline of species (Franklin 1999), or where managers of conservation reserves have a general charter 'to conserve biodiversity' (Melzer & Clarke 2003), there is little guidance. Fire is recognised as an essential management tool, but the use of this tool is in question because information is lacking on why, how, when and where to use fire. Research is required to move past general principles to something more tangible that can be applied at the property or regional level for whole suites of species.

1.2 Determinants of Landscape Fire Patterns

Fire patterns result from fires superimposed upon a landscape that is continually being subject to several processes operating over geologic and historic timescales. The present landscape is the result of many different processes acting on a series of templates, each one forming the basis for the next (Figure 1.1). For example, landforms result from processes such as tectonics, wind, water, gravity and heat acting on the geologic template. The result is a geomorphic template with the pattern defined by the physical and chemical properties of the substrate (Rogers 1997). Processes operating on this template create a landscape with variations in topography, micro-relief and edaphic conditions (e.g. soil type, depth and chemistry). Influenced by climatic factors, species assemblages evolve to form a biological template on the landscape. Pattern on this 'natural' template (*sans* people) is determined by interactions between species, and how species respond to natural disturbances such as fire, floods and cyclones. Finally, anthropogenic forces (the intentional and unintentional actions of people) act on this natural template, with patterns again defined by species responses. Thus, the existing vegetation pattern or landscape mosaic reflects the underlying templates and forces, as well as the recent history of a location, with natural disturbances and anthropogenic disturbance operating in a repeating pattern-process sequence.

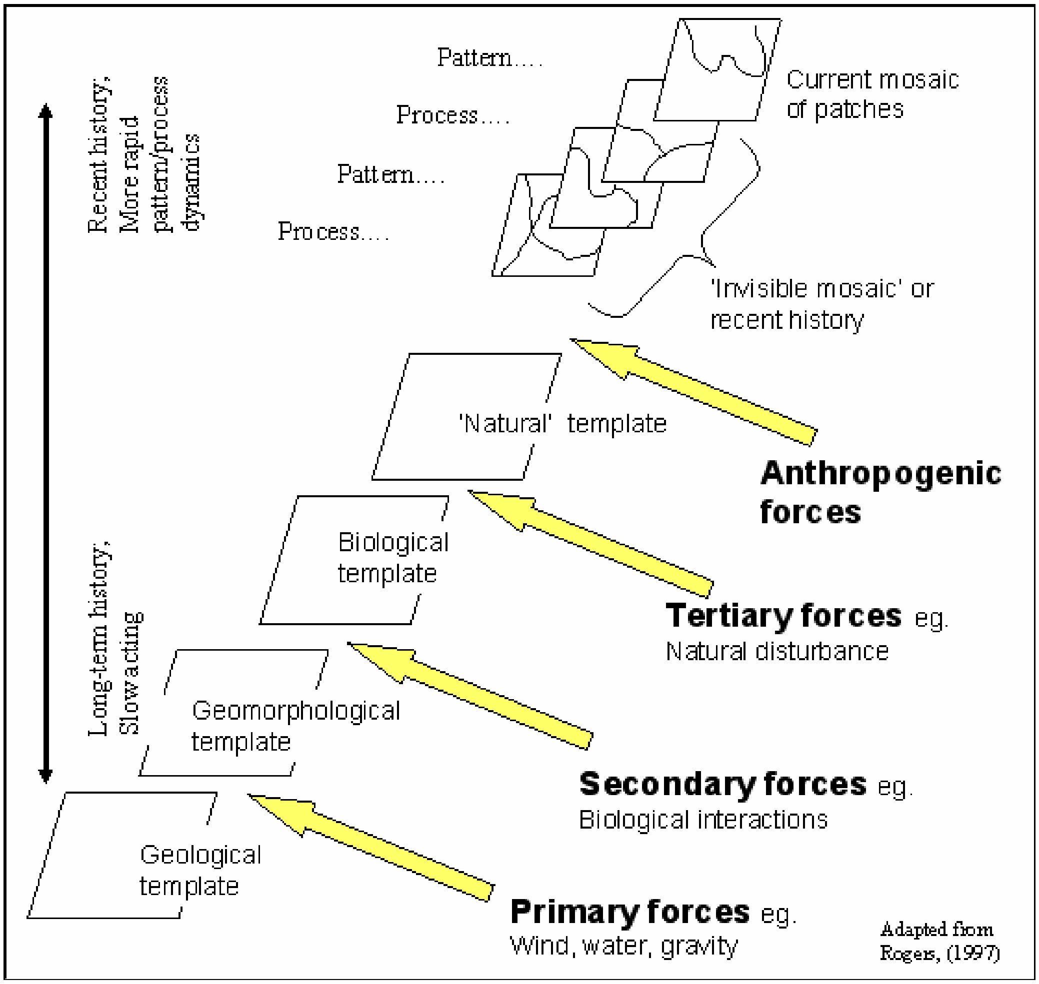

Figure 1.1 Landscape pattern is determined by different factors acting on a series of templates. The existing pattern or mosaic reflects the underlying templates and forces, as well as the recent history of a location, with natural and anthropogenic disturbances operating in a repeating pattern-process sequence.

Fire is a key driver of landscape pattern in savannas. Each fire is a discrete event that alters vegetation structure and changes resource availability and the physical environment (Turner *et al.* 2001). At the landscape scale, fire is manifested as an ongoing continuum of patchy events, with the vegetation in a state of flux as areas are burnt and recover. Fire patterns can be discerned because there is a general sequence to the changes that occur to vegetation after fire. Immediately post fire, bare ground is plentiful, but in the months and years that follow, plant communities usually change in species composition, species dominance and vegetation structure. Shoots from resprouting species are the first to appear; then with adequate soil moisture, seedlings emerge and establish. As species mature and reproduce, the open, post-fire vegetation structure becomes more complex. Depending on the vegetation type, species richness peaks in the early post-fire years or increases gradually, and then stabilizes (Gill 1999). The former response is most common in arid and semi-arid areas where species remain present, but invisible, in the soil seed bank (Gill & Bradstock 1995). Decline in species richness may occur over time because ephemeral or short-lived species mature and senesce. The cycle is repeated with subsequent fires, but there is much variation depending on fire interval, frequency, intensity, extent and timing (Gill 1977; Whelan 1995; Whelan *et al.* 2002). Thus, the species composition at any location within a landscape, at any time, is dependent on the fire history and the concomitant response of species. Other factors, such as post-fire climatic conditions, competition between seedlings, and herbivore activities, also affect the patterns apparent in the vegetation (Bond & van Wilgen 1996)

Studies of landscape fire patterns are confounded by the inter-relationships between fire behaviour, vegetation and the abiotic environment. Fire behaviour (i.e. aspects relating to ease of ignition and fire spread, shape and flame characteristics) strongly affects vegetation pattern, but the vegetation pattern in turn affects fire behaviour. This can be explained in terms of the fire line intensity equation (Byram 1959) that forms the basis of most fire behaviour studies:

$$\mathbf{I = HwR,}$$

where I = Fire intensity (rate of heat release per unit time)
H = heat yield of the fuel burnt
w = weight of available fuel
R = rate of forward spread.

The heat yield (H) is related to the type of fuel and plant species with high concentrations of oils and resins burn more intensely (Zedler 1995). Plant species, vigour and annual growth affect the weight of fuel available (w), which increases fire intensity (Luke & McArthur 1978). Vegetation height and fuel particle size, compaction and spatial arrangement all affect the rate of forward spread (R) and, thus, fire intensity (Catchpole 2002). These variables differ greatly between vegetation types. Abiotic factors such as terrain and weather are also key determinants of fire behaviour (Catchpole 2002); these factors are discussed further in later chapters.

A considerable amount of research has focused on determining how different vegetation types respond to different fire regimes, and current knowledge for each major Australian ecosystem is reviewed in *Flammable Australia: the fire regimes and biodiversity of a continent* (Bradstock *et al.* 2002). Generalisations can be drawn from these reviews, but the ability for land managers to make accurate predictions about fire responses in specific landscapes remains limited, due to strong site-specific and fire-specific factors (Whelan *et al.* 2002). There is also significant climatic variation across the continent which affects plant growth and fire behaviour, so research conducted in temperate heath or grasslands, for example, is unlikely to apply across tropical areas and *vice versa*. Fire responses vary and fire prescriptions are not necessarily portable from site to site (Williams *et al.* 1994).

The important points are that:

- Landscape fire patterns are dynamic; patterns change through time and are influenced by stochastic events and the actions of people;
- Biota and its recovery from fire over time depends on the state of the environment before and during a fire, and the propagules available for post-fire colonization (Turner *et al.* 2001);
- Landscape patterns are not uniform due to within-fire variability (fire heterogeneity) (Turner *et al.* 2001);
- Recovery after fire depends on the spatial pattern caused by disturbance (e.g. patchiness), life history attributes of the species present, and the past fire history of the site (Turner *et al.* 2001); and,
- Variability is such that after several overlapping fires have occurred, any one *point* in a fire prone landscape has a different fire history.

Identifying fire patterns at the landscape scale enables their origin to be determined and assists with understanding the effect of fires on species survival and ecosystem functioning.

1.3 The Fire Patchiness Paradigm

Previous research into fire ecology has culminated in the 'fire patchiness paradigm'. This is the view that ecological fire management should be directed towards developing and maintaining **fire regime heterogeneity** (Bradstock *et al.* 2005; Brockett *et al.* 2001; Melzer & Clarke 2003; Rose *et al.* 1999) on the basis that a range of fire frequencies, intensities and seasons over time will benefit species conservation (Gill & McCarthy 1998). There are three key reasons why this concept is important. First, it recognises that habitat heterogeneity is a major determinant of species richness (White & Harrod 1997; Wiens 1997) and that landscape heterogeneity can be manipulated through the use of fire (Woinarski *et al.* 2005). Second, the fire patchiness paradigm can be applied in a generic sense, which overcomes confusion if land managers think that conserving species requires individual fire regimes for every species in an area, or where species have opposing needs. Third, the idea accommodates our limited information base regarding the effects of fire on species and vegetation communities. Aiming for fire regime heterogeneity allows managers to develop strategies for ecological fire management despite the lack of information.

It is instructive to examine how the paradigm has developed. Most ecological research has been directed towards understanding the nature of fire and the responses of plants and animals (Andersen *et al.* 2003a; Russell-Smith *et al.* 2001; Whelan 1995; Whelan *et al.* 2002; Woinarski 1999). Some plants are killed by fire, but for others it is a catalyst for regeneration. Likewise, some fauna are killed directly by fire, or indirectly by changed habitat conditions after fire, but others thrive in post-fire habitats. Responses depend on species characteristics and the fire history of the area. Four important components of fire history were identified. These were the type of fire (surface fuels or peat), the frequency of occurrence (fire interval), fire intensity, and season, collectively termed the 'fire regime' (Gill 1975). Gill argued that it was these factors acting **over time** that affected species composition and community structure.

Concurrently, the need to reduce the complexity of interactions was recognised (Gill 1981) so research and management could focus on critical elements. Generalisations on how fires affected ecosystems were required without necessarily knowing all the details. 'Functional groups' emerged when Gill (1981) proposed that plants be classified according to their behaviour in response to fire, rather than taxonomy. Classification of fire regimes was discounted due to the extreme number of possible variations. Gill's functional categorization has been widely adopted (Bradstock *et al.* 2002) and the system has triggered the compilation of a national register of plant responses (Gill & Bradstock 1992). This register has proved useful for enhancing communication amongst managers and scientists, and facilitating predictions of fire response in different vegetation types (e.g. Gill 1999). Applying a similar functional group concept to fauna has been slow to progress, however.

The term 'mosaic' entered fire ecology parlance in the late 1970s and early 1980s, stemming from studies in central Australia (Gill 2000). 'Mosaic' refers to the spatial pattern of vegetation on the landscape and implies a patchwork effect, with each patch of vegetation having a different fire-age, or time since fire. Proactive burning of 'patches' to achieve a mosaic pattern became 'mosaic burning'. This idea converged with findings from elsewhere in ecology, where mosaic concepts were being developed to describe regeneration patterns in vegetation, without explicitly considering fire (Gill 2000). Thus, different approaches in contemporary ecology led to an understanding of the landscape in terms of mosaics (patchiness) and dynamic pattern-process interactions.

The mosaic concept appealed to 'ecological common sense' (Bradstock *et al.* 2005), and provided a means to accommodate the requirements of a broad range of plant and animal species. Since habitat diversity is critical to the maintenance of fauna diversity (Recher 1986), and habitat diversity can be generated by a diversity of fire regimes, it follows that imposing fire to create spatial and temporal variation in vegetation will provide for the diverse habitat requirements of animal species. It was also recognised that intermediate fire regimes would lead to greater landscape heterogeneity than frequent large fires or small infrequent fires, which both have homogenizing effects (White & Harrod 1997). A 'fine-scale mosaic' is considered desirable on the premise that a range of fire age classes in close proximity will provide for the greatest number of species, since habitat will be available for those that favour early, middle, late, or a combination of fire recovery stages (Bolton & Latz 1978; Fox & McKay 1989; Fraser *et al.* 2003; Masters 1993; Woinarski 1999). A direct relationship between fauna population sizes and the scale of fire mosaic is yet to be proven, however (Short & Turner 1994). Several studies investigating variation in fire regime and its effect on biota have only detected change at the burnt/unburnt level, with no differentiation with regard to season or intensity of burning (Andersen *et al.* 2003a; Mills 2004; Parr & Andersen 2006; Short & Turner 1994).

A further difficulty arises when trying to define what patch size constitutes a 'fine-scale mosaic'. This is because patchiness at a scale suitable for one species may not suit another. Patch-size suitability depends on the scale of the fire pattern relative to the home range of individual organisms (Woinarski *et al.* 2005). It also depends on whether a population perspective or individual organism perspective is being considered. If the scale of fire pattern is greater than the home range, persistence is still possible at the population level, but age to maturity, dispersion and recolonisation need to be considered (Woinarski *et al.* 2005)

Bradstock *et al*. (2005) raised concerns that the notion of fire mosaics is oversimplified because patterns are interpreted at a single point in time, or on the basis of individual fire events. It does not effectively address variability in fire interval, intensity and timing. These authors proposed that thinking in terms of fire regime heterogeneity might overcome the problem as the terminology encapsulates temporal change and diversity (Bradstock *et al*. 2005). The semantics are important. To date there is no universal definition of 'mosaic-burning'; it is applied in reference to single fire events (Gill *et al.* 2003), post-fire ages of vegetation, and holistically to include the temporal and spatial elements of fire management (a mosaic of fire regimes) (Brockett *et al.* 2001; ESA; FEWG 1999; Melzer & Clarke 2003; Russell-Smith *et al.* 2003).

Discussions on fire patchiness are also clouded by issues of scale. Patchiness can be described at two levels: that which occurs within a fire scar (intra-patch or **internal** heterogeneity) or that which relates to the configuration of fire scars (inter-patch or landscape heterogeneity) (Gill *et al.* 2003; Parr & Andersen 2006). ‘Mosaic burning’ is used in the latter sense and ‘patchy fire’ is used in the context of internal fire patchiness. ‘Patchy fires’ are an extension of the fire regime heterogeneity concept, whereby managers aim for variation in fire intensity to the extent that unburnt areas remain within the overall fire perimeter. Although the terminology is yet to be standardised, the ‘fire patchiness paradigm’ has provided useful directions for ecological fire management.

1.4 Adaptive Management and Fire Planning

The fire regime heterogeneity, or patchiness paradigm, is appealing because of its generalised nature. By assuming ‘pyrodiversity begets biodiversity’ (Parr & Andersen 2006), managers can pay less attention to the inherent variability within and between fires, our limited knowledge on the ecological effects of fire, and the potentially innumerable permutations between fire, terrain, climate, weather and biota. However, fire ecologists also espouse land managers to undertake ‘Adaptive Management’ (Andersen *et al.* 2003b; ESA; FEWG 1999; Tasker & Baker 2005), which presents a quandary. Adaptive management refers to a systematic process for improving management by learning from the outcomes of previous policies and practices (Holling 1978; http://www.for.gov.bc.ca/hfp/amhome/AMDEFS.HTM). It provides a way of managing natural resources in the face of uncertainty and variability, where the outcomes of management decisions are difficult to predict. The process is cyclic (Figure 1.2); plans for achieving management objectives are developed using current knowledge and monitoring is undertaken to track the success or failure of implemented management actions. Reviewing outcomes contributes new knowledge, which can then be used to fine-tune management plans for future implementation. The methodology makes sense intuitively, but hinges on setting explicit ecological management objectives. The problem is that the patchiness paradigm is non-specific but clear objectives are required for planning, monitoring and reporting. Fire regime heterogeneity and management actions to conserve species need to be linked at the operational level (Parr & Andersen 2006).

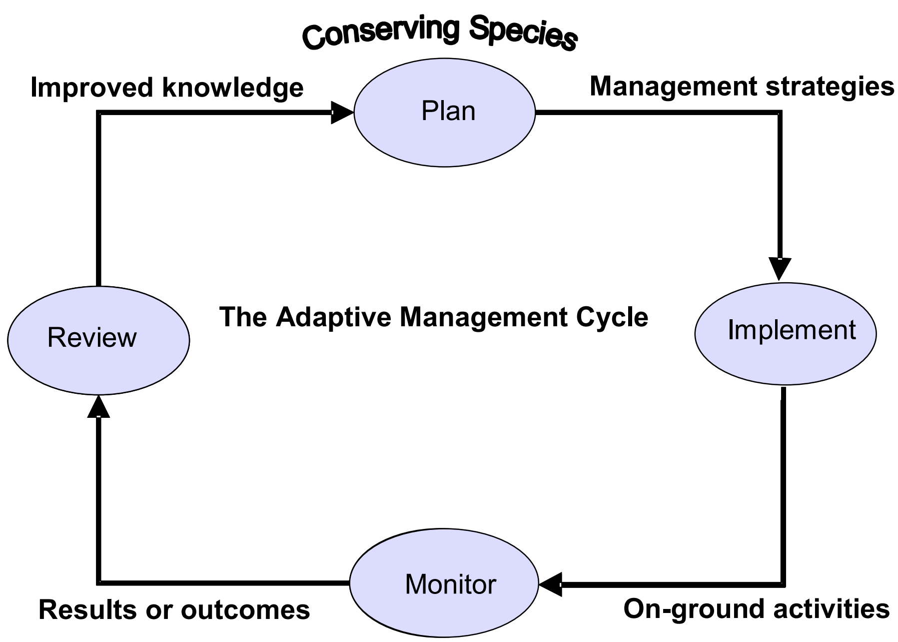

Figure 1.2 The Adaptive Management Cycle. Plans for achieving management objectives are developed using current knowledge and monitoring is undertaken to determine the outcome of management actions (based on Holling 1978).

Adaptive management accommodates the many stochastic factors that affect the response of the biota to fire, such as climatic variability or episodic events, which are beyond the control of land managers. By providing for continuous review, emphasis can be on minimising the risk of extinction, rather than maximising population densities (Tasker & Baker 2005).

Policy documents that emphasize the maintenance of vegetation at different post-fire recovery stages (ESA; FEWG 1999; Melzer & Clarke 2003) reflect adoption of the patchiness paradigm by land management agencies. However, guidance on its practical integration with adaptive management has only recently emerged. Parks Victoria (FEWG 1999) suggest using ‘key fire-response species’ as the basis for setting measurable fire management objectives for a community. These are species that are either vulnerable to too frequent fires, or to long periods of fire exclusion, which can be determined from the species fire-response. The requirements of key species indicate the upper and lower thresholds of tolerable fire frequencies for an area. From this information, models can be developed to describe the desired distribution of fire-age classes within each plant community. By combining fire history data with vegetation mapping using a geographic information system (GIS), fire-age distribution curves can be determined for particular vegetation types and management outcomes can be compared with the ideal.

In keeping with the patchiness paradigm, the FEWG’s (1999) methodology assumes that maintaining vegetation types at a range of post-fire recovery stages will provide a variety of habitats for fauna. The importance of key life history features of fauna are acknowledged, but the attributes of **plants** are used to set threshold fire-intervals. The method also centres on temporal rather than spatial variability. Because fauna can be affected by the size of fire, distance to unburnt areas, and patchiness within burnt areas, these factors need to be considered. Nonetheless, the method is a first step to incorporating fire-patchiness objectives within a management system.

A similar approach is the ‘Patch Mosaic Burning’ system used in South Africa where ‘thresholds of potential concern’ are set for specific indicators (Parr & Andersen 2006). These indicators provide upper and lower thresholds which, if reached, trigger a careful review of management practices. The method has scope for setting operational, environmental, or species-specific targets. For example, it can consider the area burnt or unburnt at different times, proportion of landscape in different stages of recovery, or the population status of individual species. Emphasis to date has been on operational achievements - links with biodiversity objectives are currently weak (Parr & Andersen 2006).

Data on the fire responses of species are required for setting objectives, but information on the regional setting, fire behaviour, climate and operational resources (ancillary data) are also required to move towards implementation (Figure 1.3; Smith *et al.* Appendix 1). The regional setting relates to the management area's position in the surrounding landscape. It includes information on typical burning patterns for the area, fire-hazard areas and fire-spread patterns. Fire behaviour information is required in order to assess the conditions for burning. From this information, potential ignition locations and times can be identified, and strategies determined to contain planned fires within defined areas.

After implementation, monitoring is a core activity for adaptive management because it enables management actions to be critically assessed and improved. Monitoring can focus on individual organisms, vegetation assemblages, entire landscapes or these factors in combination. The indicators and techniques used depend on the purpose of monitoring and each has different strengths and weaknesses (Smyth *et al.* 2003). At all levels of interest, specific and measurable management objectives are required, so that observed and expected results can be compared (FEWG 1999; Whitehead *et al.* 2001)

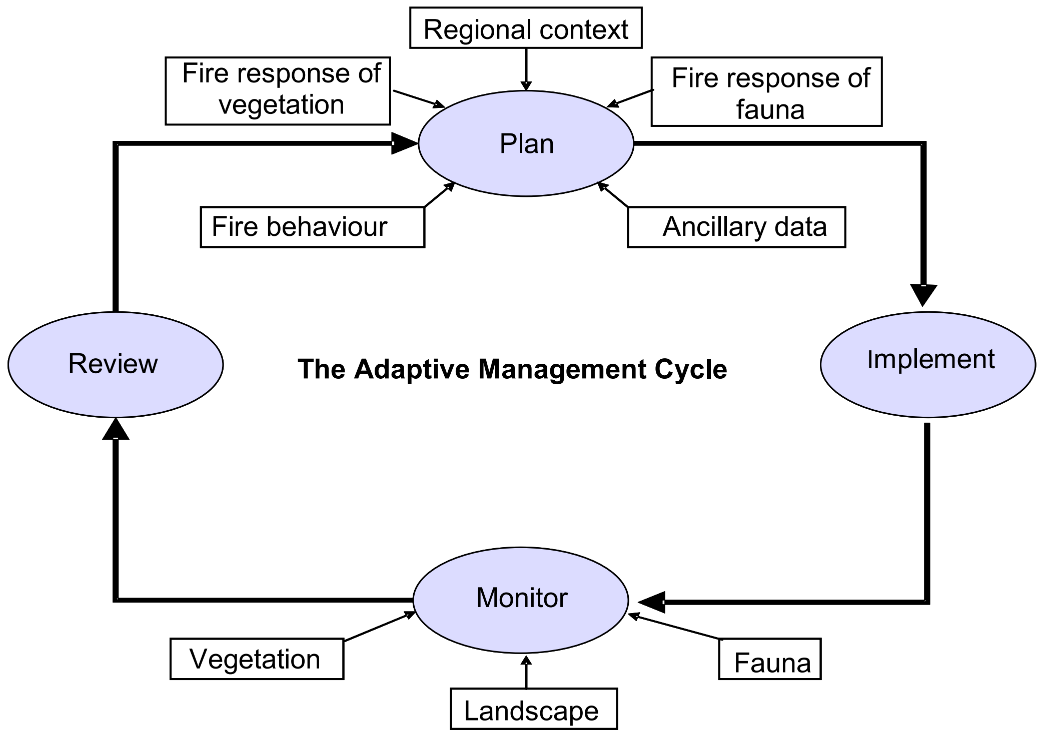

Figure 1.3 Adaptive management requires information at a number of levels. Information on the regional setting, the predicted and potential responses of flora and fauna to different fire regimes, and information on how fire behaves under different environmental conditions is required for effective planning.

1.5 Subject Matter

This thesis is concerned with fire regime heterogeneity and landscape-scale monitoring using remote sensing. Given the nexus between fire patterns, landscape patchiness and species conservation, and the ability to measure patchiness using remotely sensed data, developing remote sensing tools for monitoring biodiversity is a logical step. Two recent reviews have identified remote sensing as an essential component of a national level biodiversity monitoring system (Smyth *et al.* 2003; Whitehead *et al.* 2001). However, this technology has not been adequately tested (Smyth *et al.* 2003) and linkages to biodiversity measures need to be improved

(Whitehead *et al.* 2001). If key response species can be used to define patchiness targets, then achievement of targets can be assessed using remotely sensed data (with field work to confirm the continued presence of key species). Alternatively, application of the 'patchiness paradigm' could be assessed in its own right by measuring change in landscape pattern indices (Turner & Gardner 1991). The paradigm assumes patchiness is a surrogate for biodiversity; however, this requires confirmation. The question remains: what should we be aiming for and how will we assess it?

1.6 Aims and Significance of the Research

In this thesis, I aim to investigate options for objectively setting fire-patchiness targets and to consider how fire-patchiness can be monitored at the landscape scale using remote sensing. The intention is to demonstrate a process for informing fire management where species conservation is a desired outcome.

The first aim is to test the use of 'key fire-response species' for setting measurable fire management objectives in an area where there is a dearth of data on how species respond to fire. In Victoria, Australia, the methodology relies on relatively detailed vegetation mapping and on having comprehensive and readily accessible databases on the status and distribution of key species (FEWG 1999). Fire history information is also available throughout that State. Elsewhere, such thorough mapping and information on species distributions and fire effects is typically limited or unavailable. Species conservation outcomes might be achieved by applying the 'patchiness paradigm', except there is no universal patchiness target so specific objectives remain nebulous. A general process for setting realistic fire-patchiness objectives is required. The greatest gains would result from a process that could be applied to management units at the property scale or larger, as this is the scale at which fire management is usually undertaken.

The second aim is to explore ways to incorporate the fire-responses of fauna into the decision making process. This may help safeguard the assumption that managing vegetation heterogeneity will accommodate the needs of fauna. A third aim is to investigate spatial variability within and between fires, and improve our understanding of how best to use remotely sensed data for recording fire-related information at the landscape scale.

The area selected for the case study is the Mount Isa Inlier, a bioregion in northwest Queensland, Australia (Figure 1.4). Although there has been a significant amount of fire research in Australia's tropical savannas (Andersen *et al.* 2003a; Williams *et al.* 2002), most studies have concentrated on the more mesic savannas, not the drier semi-arid savannas. There has been no fire research specific to the Mount Isa Inlier, where fire patterns are expected to be different given its tropical semi-arid climate (Felderhof & Gillieson 2006).

The approach taken is:

1. To describe the current landscape fire patterns and identify the key drivers of this pattern;
2. To determine which key species are potential indicators of an appropriate fire regime for the most prevalent vegetation type, low open woodland dominated by spinifex grass (*Triodia* spp.) and snappy gum (*Eucalyptus leucophloia)*;
3. To investigate inter-fire variability and provide insight to landscape fire monitoring using remote sensing; and
4. To integrate this information and make suggestions for ecological fire management targets appropriate for the bioregion.

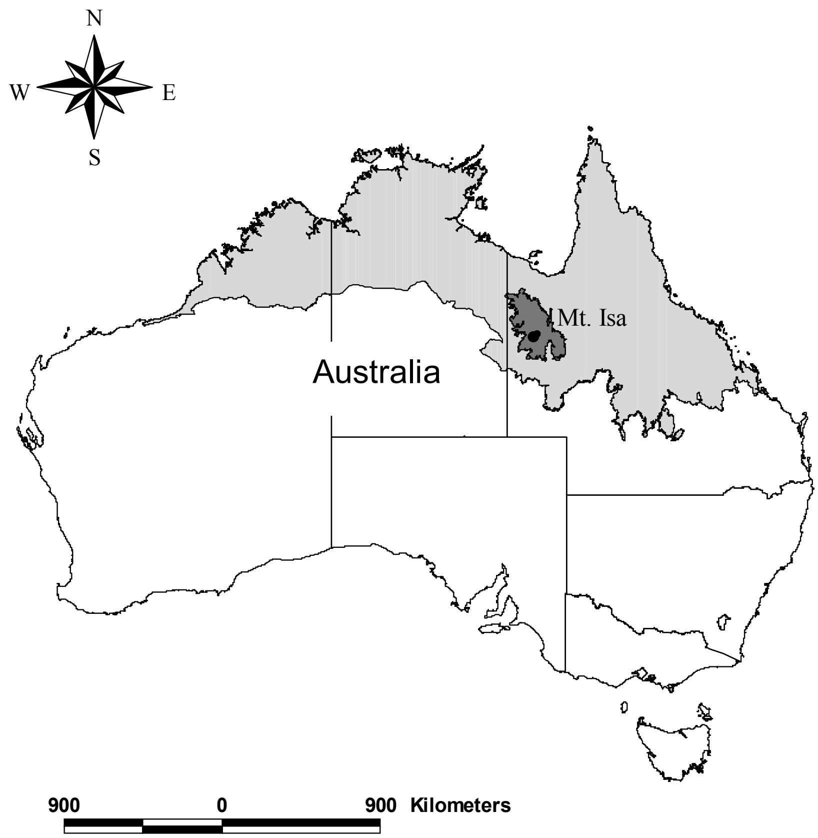

Figure 1.4 Location of the study area. The Mount Isa Inlier bioregion (dark grey) lies within Australia's tropical savannas (light grey).

1.7 Research Strategy and Thesis Structure

The research framework used in this thesis is based on the adaptive management cycle, concentrating on components that will contribute to planning and monitoring (Figure 1.5). This framework will appear at the commencement of each chapter to orientate the reader and emphasise the applied nature of the research. The early chapters focus on fire in the landscape, emphasising the interactions between fire and vegetation, inter-fire variability and the mapping and monitoring of fire patterns using satellite imagery. The fire responses of key vertebrate fauna are then considered.

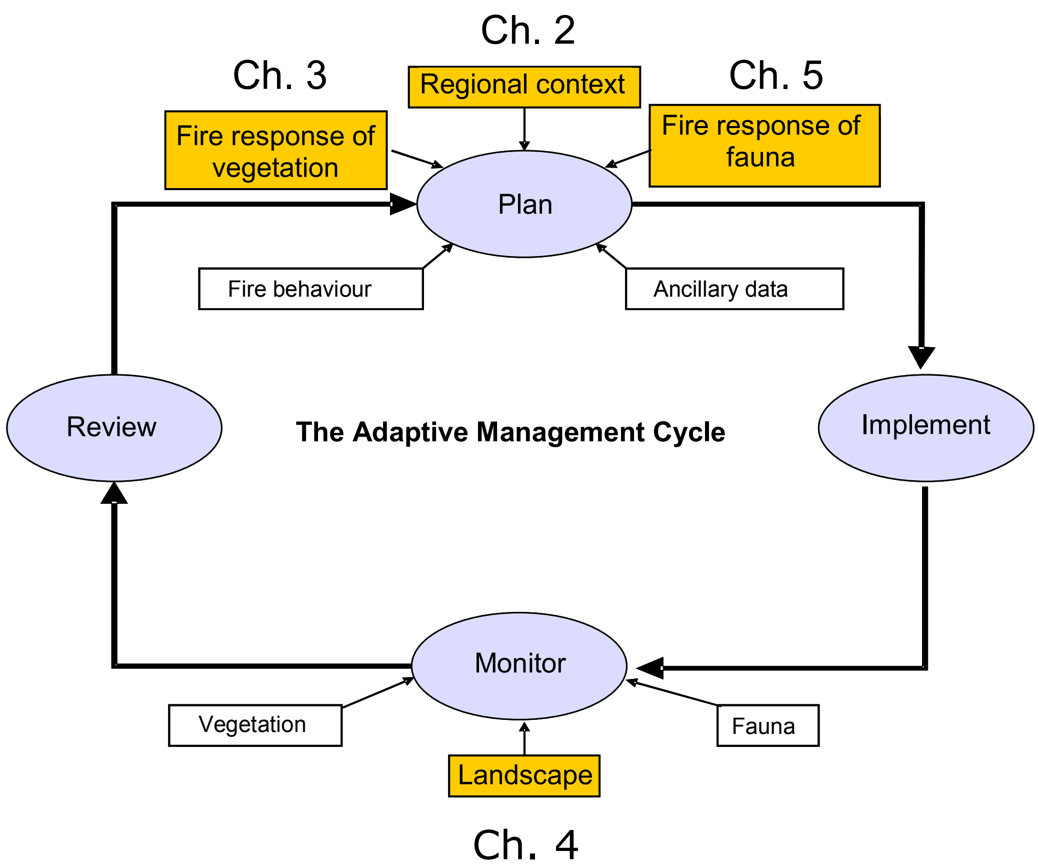

Figure 1.5 Model of the adaptive management cycle showing information requirements that will be addressed by this thesis.

Chapter 2 describes the study area and examines the fire patterns between 1998 and 2003. Satellite imagery was used to map the fire history of the Mount Isa Inlier for this period. The data are interpreted with respect to the overall climate and seasonal variability. Chapter 3 examines the response of the dominant vegetation after fire. The assessment of floristic patterns is based on changes detected over five years by surveying sites of different fire-age (time since fire). Fuel load dynamics and variability in fire severity are also examined.

The interaction between fire and landscape pattern is investigated further in Chapter 4. Fine-scale satellite imagery (1-m^2 pixels) was used to map fuel load across the landscape. Geostatistical analyses are undertaken to determine the relative importance of factors such as fire history, topographic position, proximity to drainage lines and plant species in determining fuel load. The mapped area was subsequently burnt, so fire-severity mapping is explored using satellite imagery at the same scale. Thus, landscape fire patterns are interpreted with regard to the response of individual plant species to fire, as well as in terms of landscape features and pattern-process interactions.

The question of whether the detected fire patterns are adequate for conserving vertebrate fauna is addressed in Chapter 5. Mindful that a universal prescription is impossible, Chapter 5 explores options for grouping species in order to identify those that are fire sensitive, so that fire management can be directed towards their requirements. Devising a process with broad applicability is desirable as it overcomes the problem associated with a perceived need to specify individual fire regimes for the conservation of every species in an area, which is an impossible ask. An approach that concentrates on the most sensitive species will allow more explicit ecological management objectives to be set. Three different ways of grouping species are theoretically investigated. Then, predictions are tested for one group (birds) in the field. Test results are interpreted in terms of the process applied, issues associated with setting ecological fire management objectives, and monitoring the effects of fire at the landscape scale.

Chapter 6 ties together information on the effects of fire on flora and fauna and interprets the current fire pattern for the semi-arid Mount Isa Inlier from the perspective of species conservation. The synthesis provides an overview of the fire patterns in the bioregion and the issues associated with adaptive fire management. This chapter concludes with a discussion of the implications of this research in terms of the future management of the tropical savannas in northern Australia.

2. BROAD SCALE FIRE PATTERNS[1]

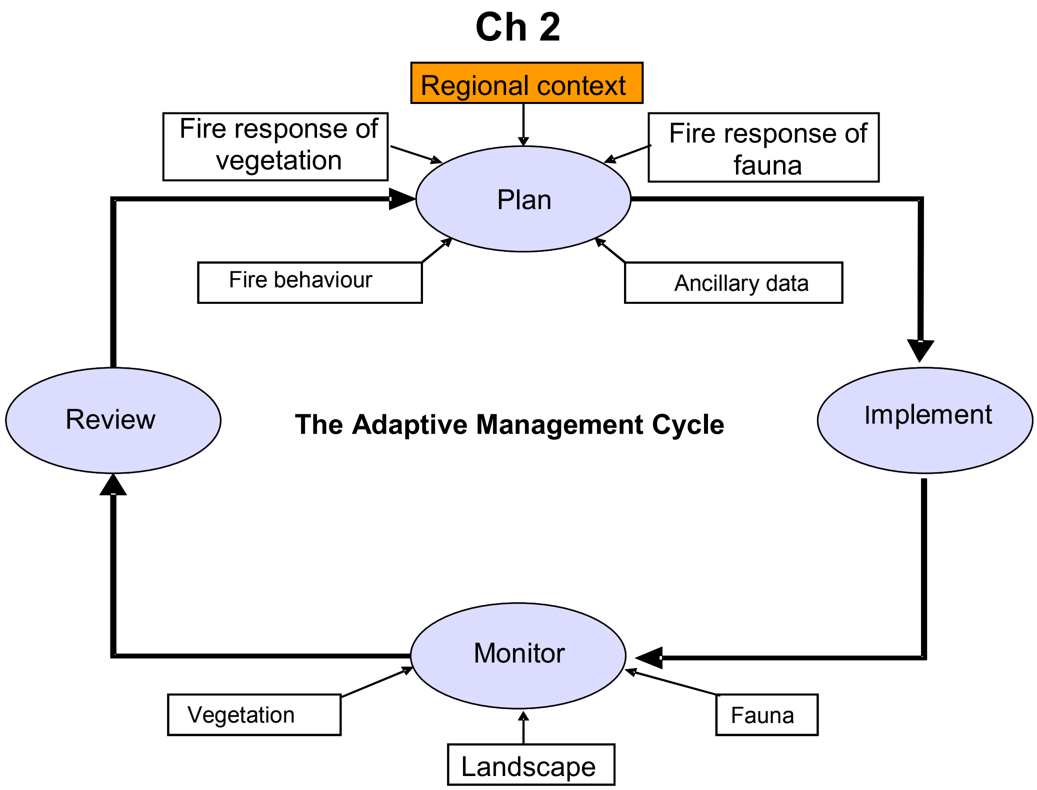

Figure 2.1 The regional setting needs to be considered when planning the use of fire. Such information enables fire management to be in context with contemporary fire patterns, allows areas to be identified where wildfire would be unacceptable, and is used to determine areas for fire ignition and containment.

2.1 Introduction

Fire-management planning at the property or regional scale requires a clear understanding of the regional landscape setting, fire history and key variables driving fire patterns in the area (Figure 2.1). This information is essential for setting management objectives and determining strategic actions to achieve them. For example, inappropriate burning patterns have been suggested as a causal factor in

[1]The majority of this chapter has been published as: Felderhof, L. and Gillieson, D. (2006) *Comparison of fire patterns and fire frequency in two tropical savanna bioregions*, Austral Ecology 31:736-746 (See Appendix 1).

the decline in native fauna (Franklin 1999; Woinarski *et al.* 2000). An examination of fauna records in conjunction with fire history maps may elucidate common elements where species have remained stable, increased or declined. Further, assessing recent fire history with respect to vegetation may highlight fire-sensitive species and communities in need of protection (Russell-Smith *et al.* 2002), or fire-dependent communities that have been long unburnt (Harrington & Sanderson 1994). Areas of wildfire risk may be identified from previous patterns of repeated fires or by locating long-unburnt areas in fire prone vegetation. Once fire-management objectives are established, the regional context is pertinent to matching the conditions for burning with the desired outcomes. Understanding the prevailing landscape fire patterns helps to determine when and where fires might spread and where they can be contained.

Satellite imagery makes it possible to record recent fire history over large areas. Remotely sensed satellite data provide a spatial representation of burnt and unburnt country over time, with burnt areas termed 'fire scars' (Dyer *et al.* 2001). These data can be collected over time to determine the extent and timing of fires on an annual basis or provide insight to fire frequencies and fire return intervals at different points in the landscape (Allan *et al.* 2001; Allan & Southgate 2002; Russell-Smith *et al.* 2000; Smith *et al.* 2000). Results can be used to investigate long-term trends in fire pattern and relate these to climate data (Allan *et al.* 2003; Allan & Southgate 2002). Fire patterns can also be linked to other spatial data using a geographic information system (GIS) to investigate relationships between species or vegetation communities and fire (Russell-Smith 2002a). Analysis at the regional scale is particularly advantageous as it corresponds with the landscape scale at which fire management is undertaken (Rose *et al.* 1999; Turner *et al.* 2001).

Mapping fires from satellite imagery has been undertaken in Australia since the 1980's (Press 1988). The approach has been used to compare the extent of fire in different land management systems and for broad scale interpretation of the fire regimes operating in mesic savannas (Allan & Southgate 2002; Gill *et al.* 2000; Russell-Smith *et al.* 2002; Russell-Smith *et al.* 1997b; Russell-Smith *et al.* 1998). It has also been combined with ground-based data to compare fire regimes at two ecologically similar national parks (Edwards *et al.* 2001). Although Russell-Smith *et al*. (2000, 2003) compared burning patterns between vegetation types and across jurisdictions of Australia's tropical savannas, Arnhem Land in the Northern Territory has been the main focus of fire research. Very little assessment has been carried out in semi-arid Australian savannas, particularly in Queensland. Studies in Queensland have been limited to Cape York Peninsula in the northeast, where fire scar maps have been validated by comparing mapped images with field data (Collett *et al.* 2001). Fire mapping has been used for planning fire management on Cape York Peninsula (Thompson 2003).

The aim of this chapter is to determine the recent fire history of the Mount Isa Inlier, a tropical, semi-arid bioregion in northwest Queensland. Mapping was undertaken at the bioregional scale because bioregions are, by definition, areas with similar landform, vegetation and climate (Thackway & Cresswell 1995). These three factors are strong determinants of fire occurrence. Elucidating fire pattern, and determining how it differs from more mesic savanna areas, provides a regional context for planning the use of fire. Findings here are compared with general fire patterns reported for tropical savannas from different regions and time periods (Williams *et al.* 2002).

2.2 Methods

2.2.1 The Study Area

The Mount Isa Inlier covers 7,607,464 ha and is located in Australia's tropical savannas (Figure 1.4). The climate is characterised by low and highly variable annual rainfall and warm to hot daily temperatures. The mean annual rainfall is 480 mm, which falls predominantly in the wet season between December and March (Clewett *et al.* 2003). The strongly seasonal, wet-dry pattern is predictable but the magnitude and duration of the wet season can vary considerably from year to year. The Mount Isa Inlier can receive less than half the average rainfall in a drought year, but more than twice the average if a tropical cyclone in the Gulf of Carpentaria deteriorates into rain depression (BOM 2003). A rainfall gradient extends from east to west and from north to south, with slightly higher rainfall in the east and north respectively (Figure 2.2). The annual temperature range (difference between the summer maximum and winter minimum temperatures) increases from north to south (Figure 2.3).

The Mount Isa Inlier is characterised by rugged, rocky hills and poor shallow soil. The vegetation reflects the complex geology, infertile soil and semi-arid climate (Figure 2.4). Low open woodland with *Eucalyptus, Corymbia* and *Acacia* species predominates, with a ground layer of *Triodia* species (spinifex) (Figure 2.5). Where the soil has greater sand and clay content, *Heteropogon*, *Aristida*, *Mnesithea*, *Themeda* and *Chrysopogon* are common understorey grass genera (Gillison 1994, Sattler & Williams 1999). Rangeland grazing and mining are the major land uses. Grazing intensity is comparatively low and mining activities are locally concentrated, so natural conditions are relatively intact overall.

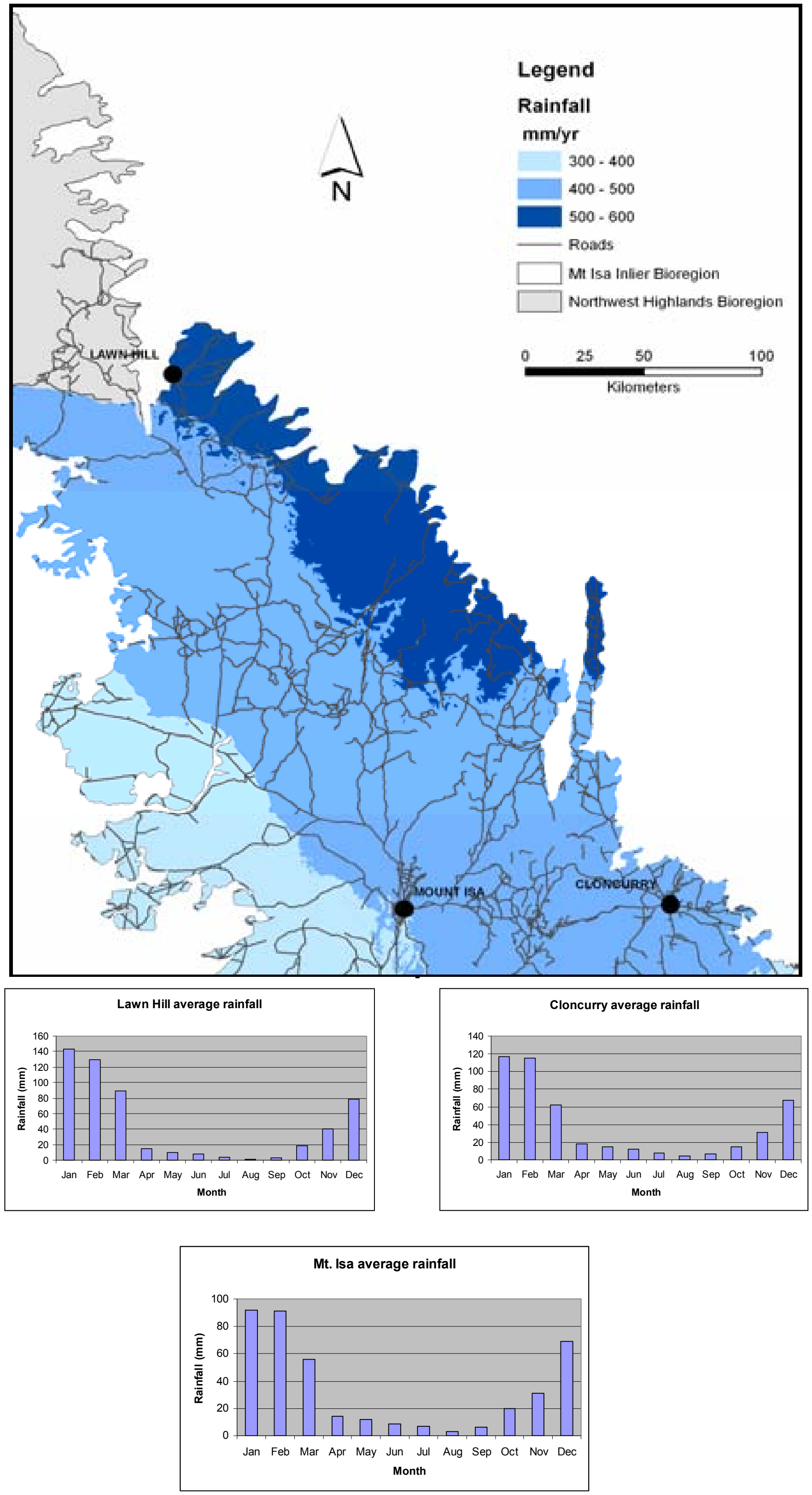

Figure 2.2 Average annual rainfall for the Mount Isa Inlier. Map based on Bioclim data (EPA 2003); bar graphs based on Clewett *et al.* (2003).

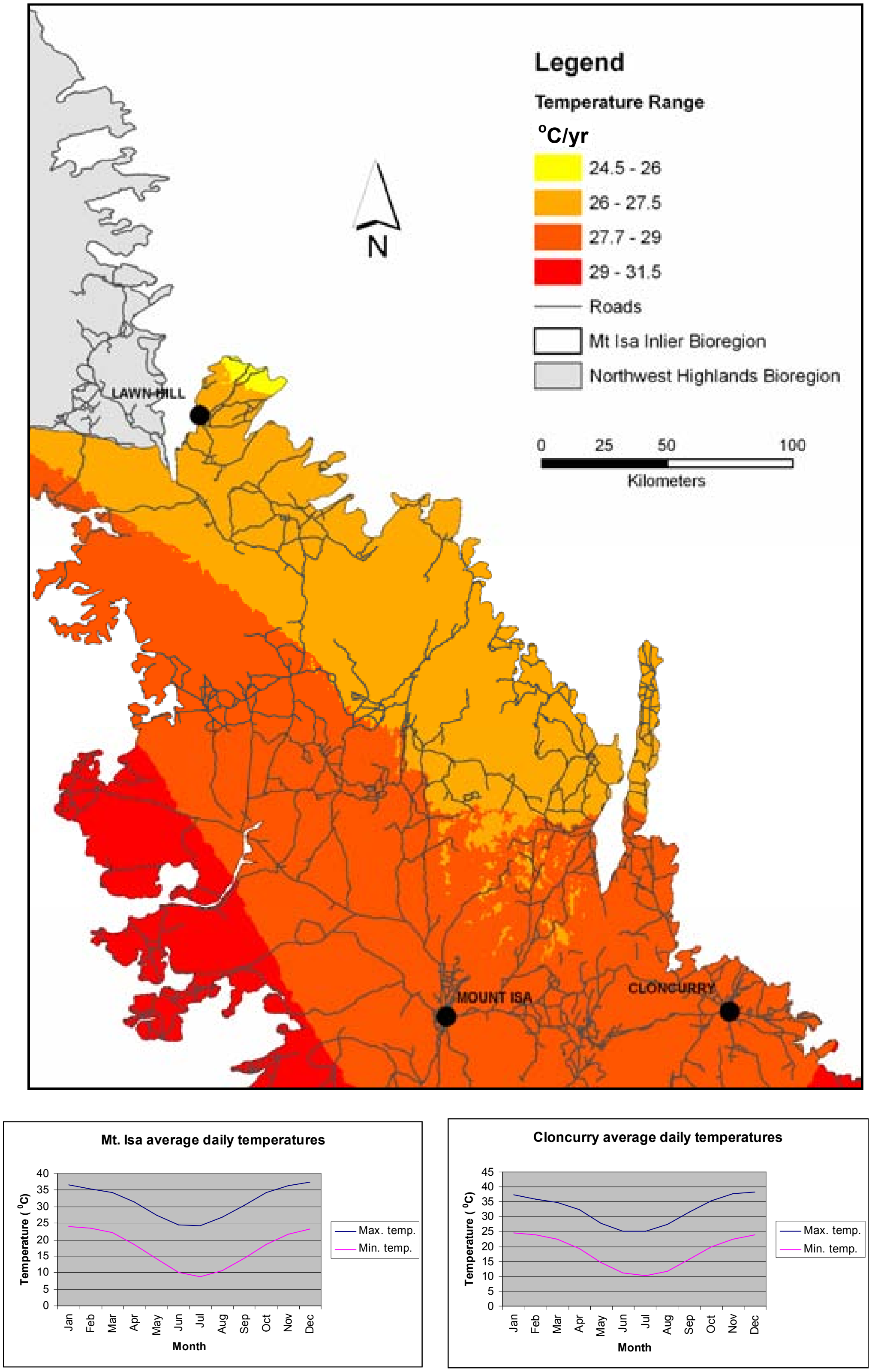

Figure 2.3 Annual temperature range (difference between the summer maximum and winter minimum) and the average annual temperatures in the Mount Isa Inlier. Map based on Bioclim data (EPA 2003); line graphs based on Clewett *et al.* (2003).

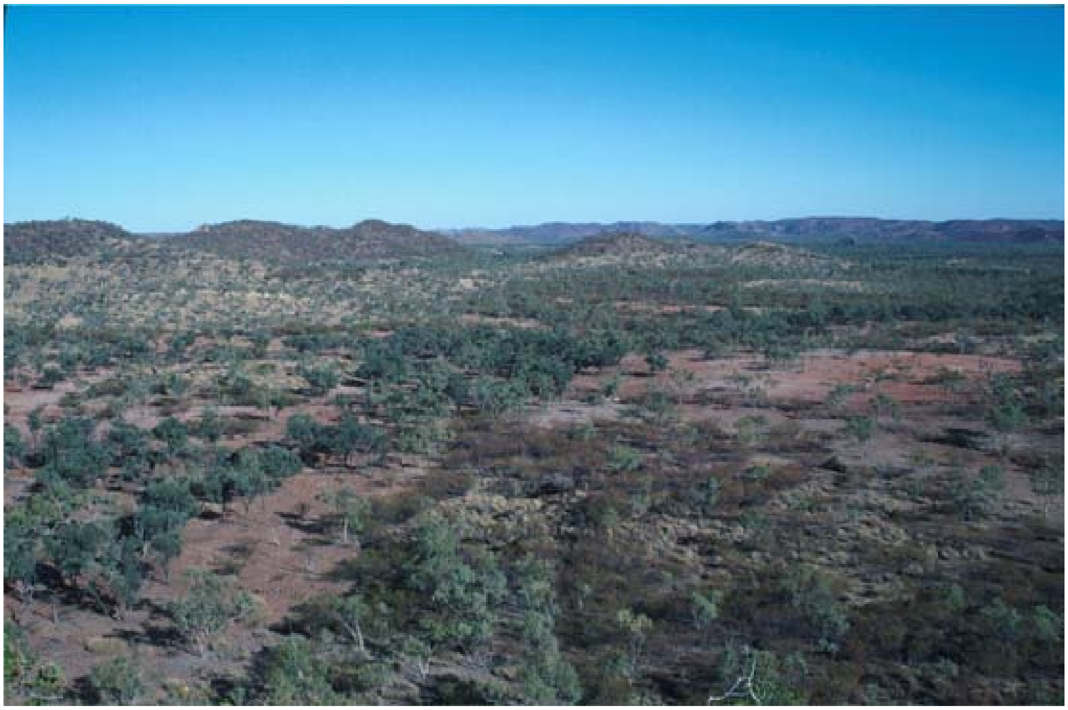

Figure 2.4 Typical landscape within the Mount Isa Inlier. The predominant vegetation is low open woodland with *Eucalyptus* spp., *Acacia* spp. and an understorey of *Triodia* spp. Other grasses occur in areas of higher fertility.

Figure 2.5 This study focussed on low open woodland of *Eucalyptus leucophloia* (snappy gum) with an understorey of *Triodia* spp. (spinifex) in various stages of recovery post fire. Plate (A) long-unburnt vegetation; plate (B) vegetation at three different fire-ages.

2.2.2 Mapping Techniques and Analysis

Fire scar mapping

Satellite imagery was used to map fire scars from 1998 until 2003. Time constraints prevented a wider date range. Most of the mapping used relatively coarse resolution Landsat 'quicklooks' (approx. 250-m x 250-m pixels) in conjunction with NOAA-AVHRR derived 'hot spots' (Craig *et al.* 2002; Smith *et al.* 2000). 'Quicklooks' are jpeg-compressed images that can be freely downloaded from the Internet. They are colour composites commonly used by potential buyers to see if cloud cover is a problem on a Landsat scene. The cost of Landsat imagery limits its use over broad areas and multiple timeframes, so as an alternative, the jpeg images were registered and an on-screen rectification process was used to provide their correct geographic location. The quicklooks were obtained from GeoScience Australia (http://www.ga.gov.au) by searching for suitable cloud free images over the study area. Once obtained, quicklooks were registered using ArcView™ GIS software (Environmental Systems Research Institute, Redlands, USA) to match visibly distinct features on the image with corresponding points on 1:250,000 scale drainage data. The intention was to obtain monthly coverage of the entire bioregion but summer monsoons meant cloud free images were scarce between December and April. Gaps in monthly coverage did not prevent fire scar identification, however, as fire scars were visible for more than 3 months and there was sufficient imagery available within that timeframe. Nearly sixty-eight percent (67.7%) of the bioregion was mapped for this study (5,149,826 ha).

Fire scars were visible as black, dark purple or dark red areas on the imagery and were mapped by eye in conjunction with NOAA-AVHRR hotspot data (Craig *et al.* 2002; Smith *et al.* 2000). The hotspot data identifies areas of active fires within 1-km^2 pixels and are available daily. The fire scar and hotspot data were used together to increase confidence that dark areas were fires, and not dark areas of bare ground or geological features (Figure 2.6). Hotspots alone were considered inadequate as ‘false’ hotspots are sometimes detected (Smith *et al.* 2000). Clusters of hotspots over areas of imagery that had become dark between one scene and another clearly indicated burnt country and provided information on the date of the fire. Fire scars were mapped progressively by month from 1998 to mid-2003 using a manual on-screen digitisation process (ArcView™ GIS software). For the last six months of 2003, data were derived from MODIS satellite sensors (250-m x 250-m pixel resolution, NASA, Washington, USA), as Landsat-TM was no longer available. The MODIS-based fire scar maps were provided by the Cape York Peninsula Development Association and were generated based on the same mapping principles, but using a semi-automated process.

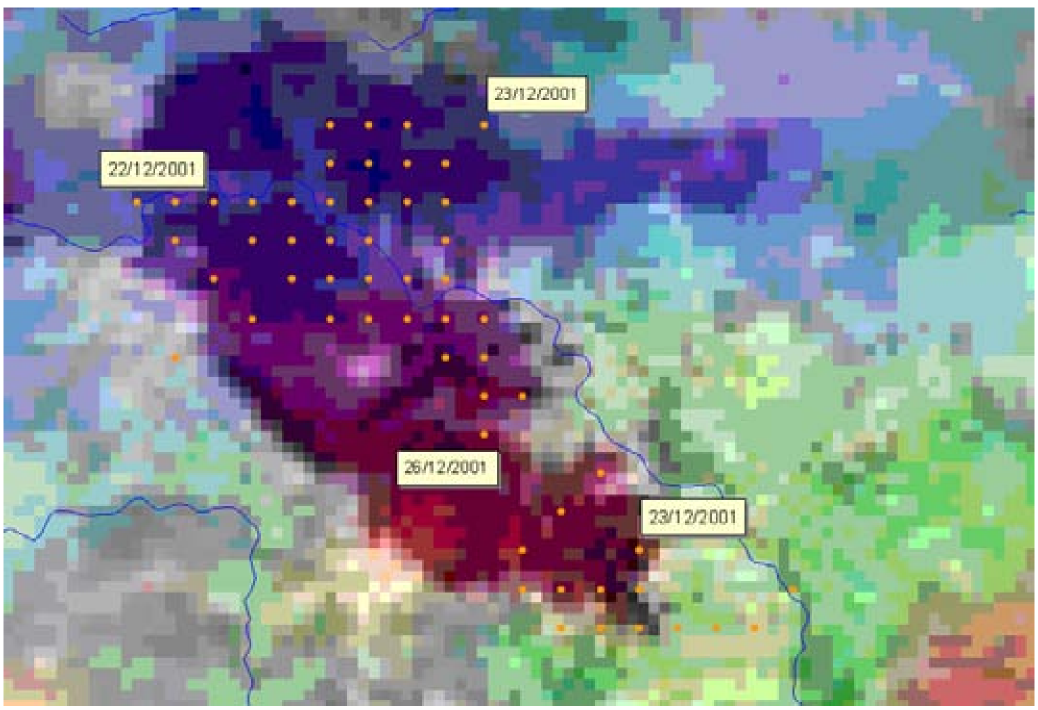

Figure 2.6 Fire scars appear as dark, black or purple areas on Landsat ‘quicklooks’. ‘Hot spot’ data from the NOAA-AVHRR satellite sensors (shown here in orange) help to confirm that these areas were burnt.

Data verification

A qualitative assessment of mapping accuracy was obtained by preparing maps at the property scale for discussion with relevant landholders. Direct consultation enabled the fire history to be checked in localised areas. Verification of fire occurrence and information on ignition source and fire behaviour was provided. Landholders were able to identify fires they had lit, incidents of arson, encroachments from neighbours, lightning ignitions and direction of fire spread.

Extensive field trips allowed for on-ground sampling of fire scars. Field inspections were conducted between March and August 2004. Apart from logistical considerations, this was considered adequate as fire patterns are preserved for three to four years after a fire where spinifex is the dominant ground cover. This occurs because there is a distinct flush of vegetative growth each wet season, which ceases with the onset of the dry, resulting in a visibly staged response.

Previous fire scar analysis of northern Australia's savannas has used coarser resolution NOAA-AVHRR data with 1-km^2 pixels (Russell-Smith *et al.* 2000; Williams *et al.* 2002) or finer resolution and costly Landsat_TM data with 30-m x 30-m pixels (Edwards *et al.* 2001). The satellite data provide reflectance values for different wavelengths of visible light and infrared radiation, which can be manipulated to enhance different features of the Earth's surface. These data processing possibilities, and finer resolution in the case of Landsat_TM, allow for detailed analysis and presumably greater mapping accuracy. To check whether mapping from quicklooks was comparable to maps derived from Landsat_TM, ten scenes of actual Landsat_TM imagery were obtained for the area. Half of the images were recorded in July/August 1999 and half were from June/July 2001. Fire scars were mapped as above and matched with the corresponding quicklook scars, resulting in 24 paired fire scars. Two GIS layers were compiled, one with fire scars mapped from quicklooks and the other with scars mapped from actual imagery. Accuracy of the quicklook mapping was assessed in two ways. First, 100 random points were assigned to each fire scar mapped from the 'actual' imagery. The number of points that also intersected quicklook scars was determined to provide a measure of spatial accuracy. The process was repeated 10 times, generating a different set of 100 points for each 'actual' polygon (fire scar) each time. Second, fire scar pairs were given unique identifiers to compare the areal extents mapped from the two types of imagery.

Data analysis

The area burnt was graphed to compare the areal extent of fire each year, as well as the percentage of the landscape burnt. Seasonal patterns were determined by plotting the area burnt per month sequentially over six years, and by pooling the data and plotting the total area burnt each calendar month.

Rainfall data were obtained from 'Rainman' software (Clewett *et al.* 2003) and used to investigate relationships between the extent of fire and rainfall. Biogeographic regions are partly based on dominant climate attributes (Thackway & Cresswell 1995), so there are broad consistencies within a bioregion. There is variation at a smaller scale, however. To accommodate the rainfall gradients within the Mount Isa Inlier, the average rainfall was determined by triangulation using data from three different weather stations. These were Mt. Isa, Calton Hills and Lawn Hill. The area burnt each year was compared with corresponding rainfall records, as well as rainfall in the preceding twelve months. Average monthly rainfall was compared with the area burnt per month on a sequential basis over the six-year time frame, and long-term monthly rainfall averages were graphed with pooled fire scar data.

Rain directly affects whether a fire will be sustained from an ignition point or not, but also acts indirectly by its influence on fuel load. The amount of rain *per se* is less significant than effective rainfall, with steady rain over a few days providing better growing conditions than the same amount in a torrential downpour (Partridge 1999). This rationale has been incorporated into the pasture growth model Aussie GRASS, compiled to predict pasture growth (fuel) under different climatic conditions (Carter *et al.* 2003). Fire scar mapping results for the Mount Isa Inlier were tested against output from Aussie GRASS (John Carter, unpublished data, 2004) to assess whether pasture growth predictions were a better indicator of wildfire hazard than rainfall records alone. Finally, fire scar data were analysed using ArcView™ GIS software to investigate the percentage of the landscape burnt once, twice or more frequently (fire frequency), and determine the resulting landscape fire age (time-since-fire).

2.3 Results

2.3.1 Data Verification

Field inspections and discussions with landholders supported the mapping results. Fire scars were always present in the areas indicated by the quicklook maps, although exact fire scar boundaries were less accurate. The total area mapped from quicklooks was very similar to that mapped from Landsat imagery; however, there were variations on an individual fire scar basis (Figure 2.7). There was significant correlation between quicklook and Landsat fire scar mapping (R^2 = 0.948, n = 29, p ≤ 0.001), with variation due to the different time of imaging with respect to the date of fire. If the time between images was less than 31 days and the fire was less than 30 days old, then quicklook mapping accounted for 99.8% $\pm$ 11.44% of the Landsat mapped area. In terms of spatial accuracy, quicklook fire scars consistently captured close to 75% of random points allocated to individual Landsat polygons (n = 10, mean 74.8%, SD 0.51).

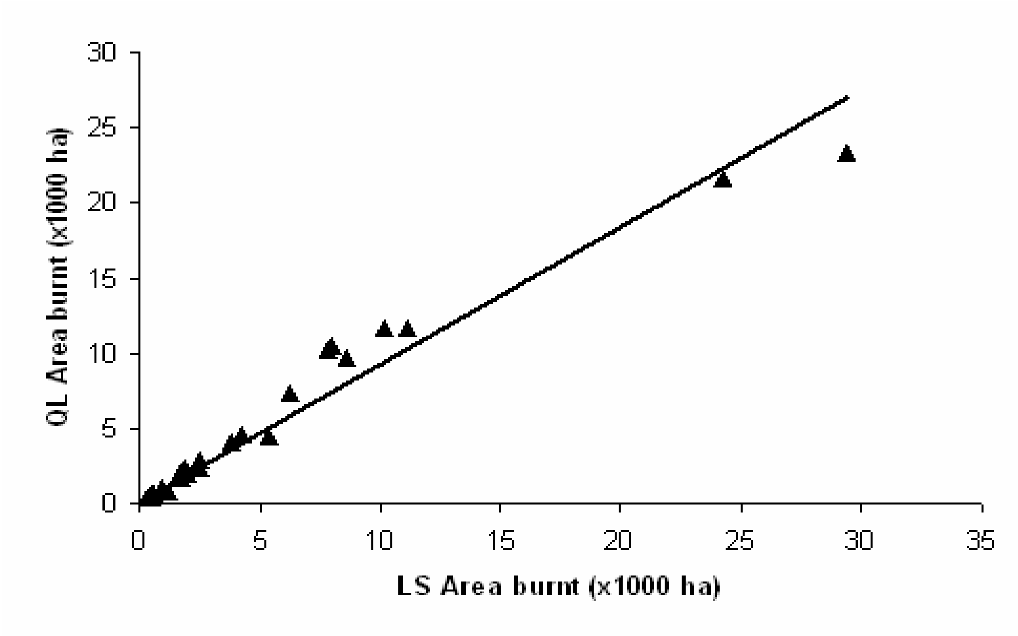

Figure 2.7 The relationship between the areas of individual fire scars mapped from Quicklooks (QL) compared with the corresponding fire scars mapped from actual Landsat imagery (LS). Y = 0.92, R^2 = 0.95, n = 29, p ≤ 0.0001.

2.3.2 Area Burnt and Seasonal Patterns

The largest area burnt in one year was in 2001 (Figure 2.8). In this year, 1,770,771 ha were burnt, representing 34.5% of the landscape. The smallest percentage of the landscape burnt in one year was 1.6% in 2003. On average, less than 10.1% of the region burnt in most years.

Fire patterns were strongly seasonal. Fires commenced around August and the area burnt increased monthly as the dry season progressed (Figure 2.9). The area burnt per month diminished with the onset of wet season rains around December. Burning continued to a lesser extent over the wet season and almost no fires occurred in the early dry season (April-July). This general pattern was observed each year, however, the commencement of the fire season, area burnt and onset of summer rains varied from year to year (Figure 2.10).

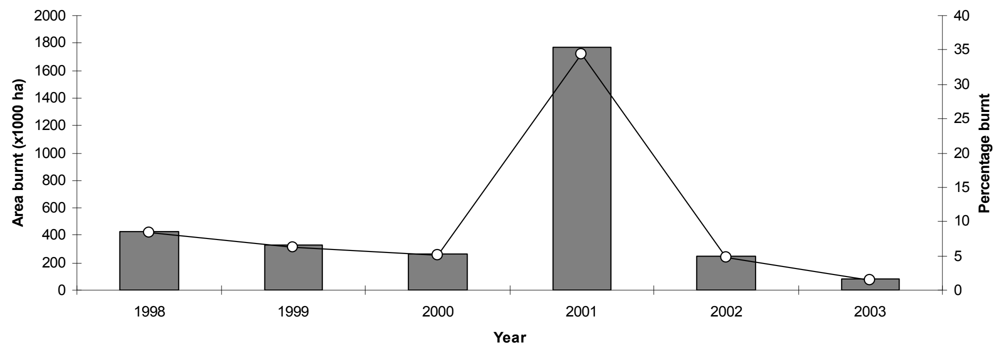

Figure 2.8 Variation in the area burnt per annum 1998-2003, as total number of hectares burnt (bars) and as a percentage of the study area (circles).

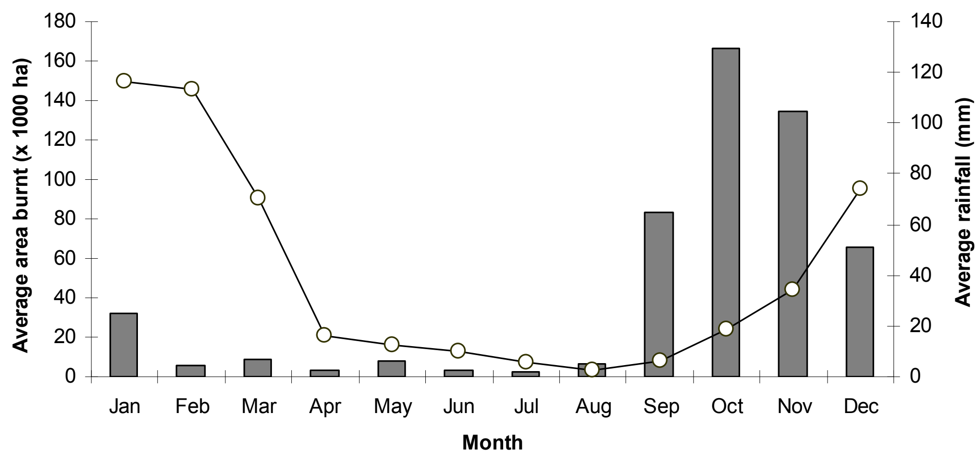

Figure 2.9 Relationship between the average area burnt per month (bars) and average monthly rainfall (lines), 1998-2003.

There was a strong relationship between the area burnt per year and rainfall in the preceding twelve months but not with rainfall during the year of burning (Figure 2.11). This relationship held when the apparent outlier (the large area burnt in 2001) was removed. R^2 values were then 0.77 and 0.146 respectively ($n = 5$, $p \leq 0.05$). The relationship between area burnt and modelled pasture growth in the preceding twelve months was even stronger (Figure 2.12).

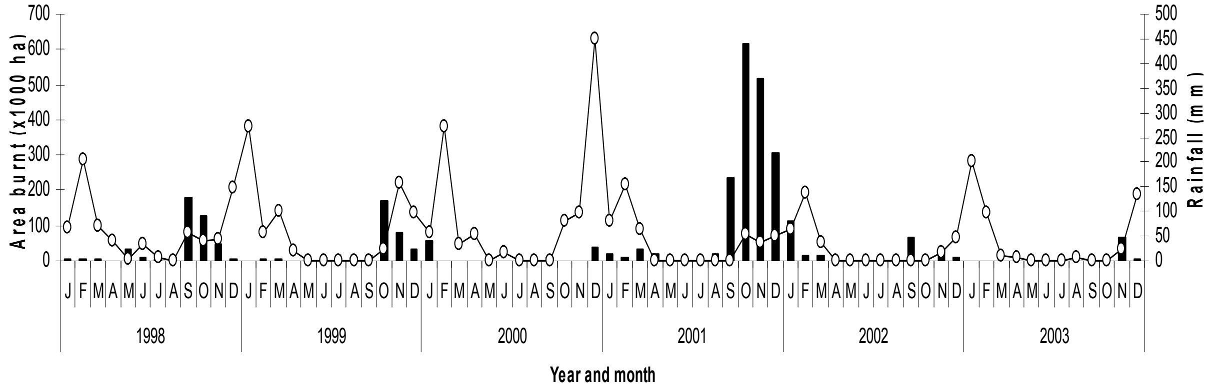

Figure 2.10 The area burnt per month (bars) and rainfall (lines) for the Mount Isa Inlier, 1998-2003.

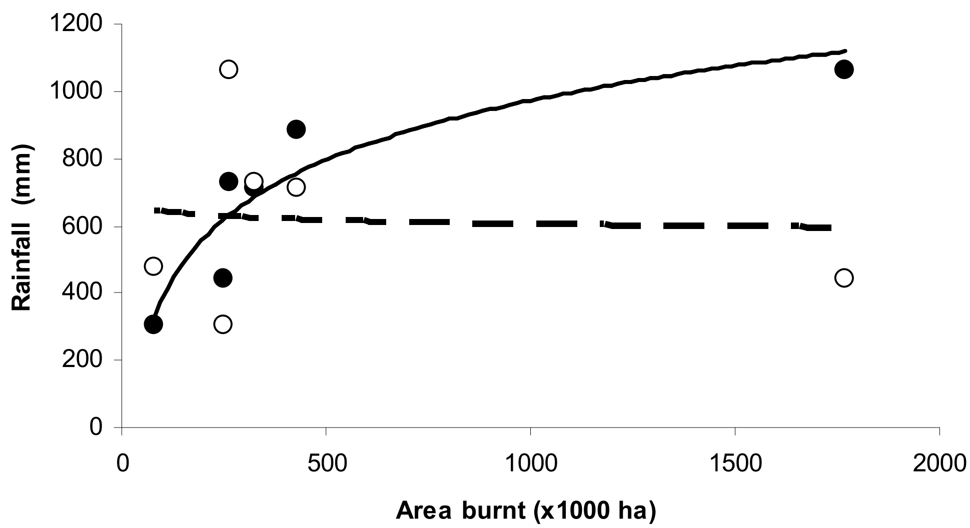

Figure 2.11 There was a strong relationship between the area burnt per year and rainfall in the preceding 12 months (solid line, R^2 = 0.84, n = 6, p ≤ 0.01) but no relationship between area burnt and rainfall during the year of burning (dashed line, R^2 = 0.068, p ≤ 0.1).

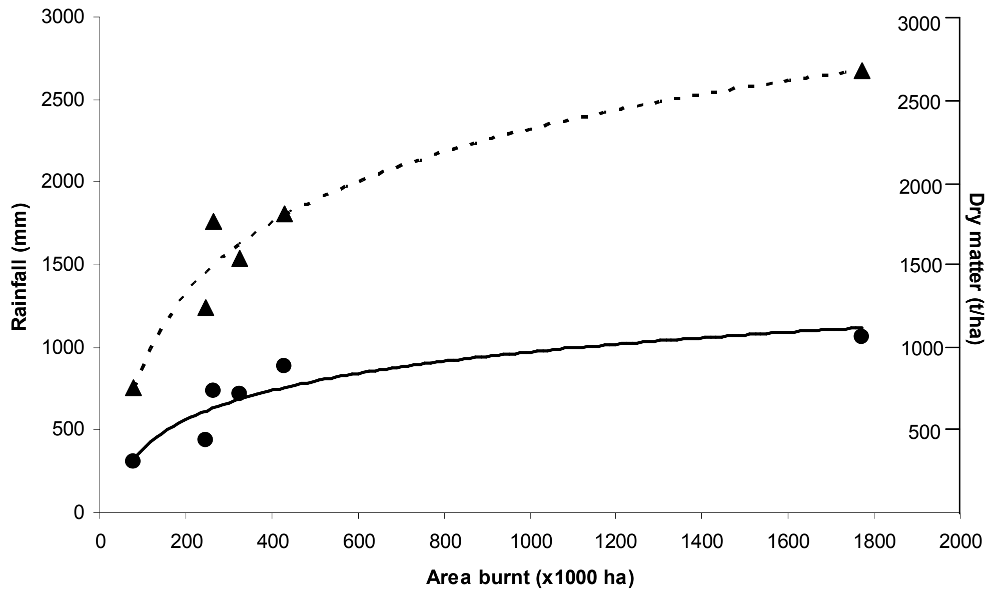

Figure 2.12 There was a stronger relationship between the area burnt per year and pasture growth (triangles and dashed line, $R^2 = 0.94$, $n = 6$, $p \leq 0.01$) than between the area burnt and rainfall in the preceding 12 months (circles and solid line, $R^2 = 0.84$, $n = 6$, $p \leq 0.01$).

2.3.3 Fire Frequency and Time Since Fire

Approximately half of the Mount Isa Inlier (45.5%) did not burn between 1998 and 2003, 48.5% was burnt once and only 5.9% burnt twice over the 6 years. The areal extent of annual burning affects landscape fire age, or ‘time since fire’. By the end of 2003, 45.5% of the Mount Isa landscape had a fire age greater than 6 years, 1.6% had burnt in the previous 12 months and 3.8 % had burnt in the preceding 24 months (Figure 2.13).

The small proportion of the landscape that burnt more than once over the six-year timeframe confirmed that fire is unlikely to spread to areas unless four or five years have elapsed since the previous fire. Of course, this depends on conditions at the time of burning, and some areas burnt in 1998 burnt again in a 2001 wildfire. The time lag between an area burning and recovering sufficiently to support a subsequent fire resulted in a patchy or mosaic landscape. Fire scars fitted together like puzzle pieces (Figure 2.14) as new fires abutted areas burnt in the previous few years.

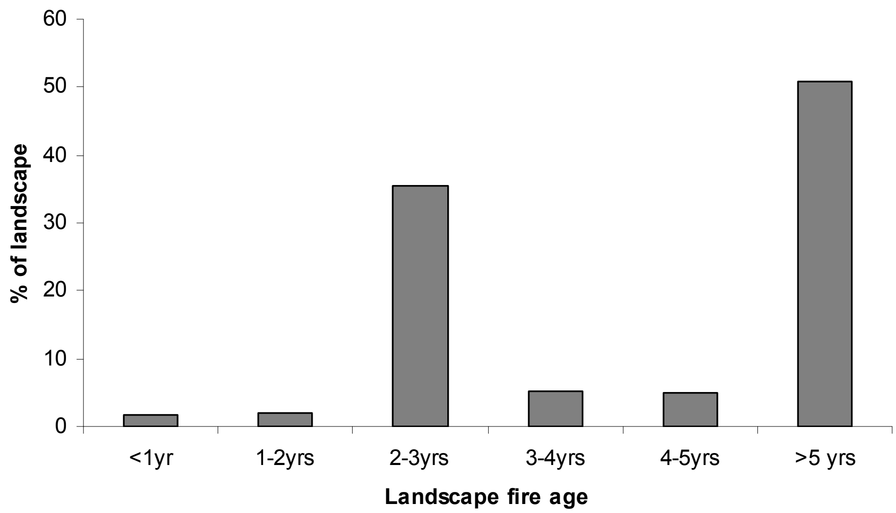

Figure 2.13 The ‘age’ of the landscape with respect to time since fire by the end of 2003. Wildfires in 2001 resulted in a large area of similar fire age.

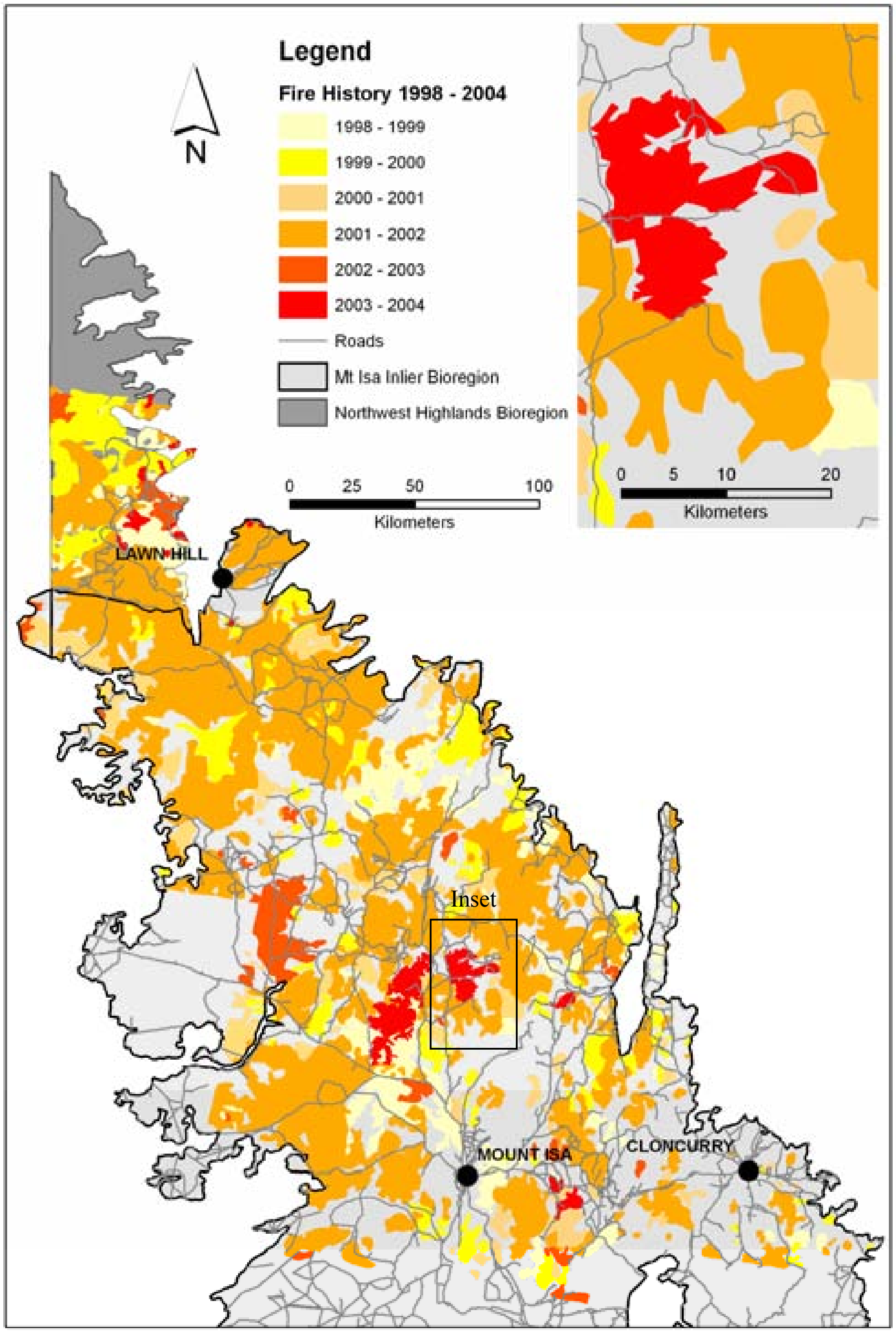

Figure 2.14 The time lag between an area burning and recovering sufficiently to support a subsequent fire resulted in a patchy landscape. Fire reliably abutted areas burnt in the previous few years (see inset).

2.4 Discussion

2.4.1 Mapping Accuracy

Freely available Landsat quicklooks were used, rather than purchased Landsat or NOAA-AVHRR imagery; this provided reasonably accurate fire mapping without the expense. Yates and Russell-Smith (2002) evaluated Landsat mapping of fire scars at different field sites across the tropical savannas and found that accuracy ranged from 84-99%. I anticipated quicklook mapping would be less accurate, but preliminary work elsewhere in Queensland suggested the method was suitable for mapping at the regional scale (P. Thompson, pers. comm., 2003). My findings suggest that the total percent area mapped using either quicklooks or Landsat was very similar, although there was some variation when individual fire scars were compared. Quicklook fire scars were not consistently larger or smaller than their Landsat counterparts and individual fire scar size discrepancies could be partially explained by considering image dates. When quicklook images were simply coarser-resolution replicates of the purchased Landsat scene, the similarity in area mapped was close to 100%. If the images did not match directly, and the dates differed by a number of weeks, I found that fire scar boundaries were less distinct and were difficult to map accurately. I also found that extreme mismatches were due to fire timing. In some cases, fires were still burning when the image was taken and fire scars had extended by the time they were captured on a following scene. The result was a larger fire scar being mapped using the quicklook or *vice versa*.

Accuracy assessments could be improved by mapping from quicklook and Landsat scenes that corresponded temporally; however, the availability of actual imagery was opportunistic and occurred after the quicklook maps had been prepared. Confidence was significantly enhanced by field inspection and landholder consultation, which suggested that precise boundary location was a greater issue than whether an area had burnt or not. I found that geographic placement was 75%

accurate, which largely reflected errors associated with the on-screen rectification process, as well as indistinct fire scar boundaries from low intensity, patchy fires, or from ageing fire scars where vegetation had started to recover. Overall, quicklooks were considered adequate for recording fire history data at the bioregional scale, as fire occurrence, timing and approximate size could be determined. Demonstration of this quicklook method is important because Landsat coverage of the continent is available back to the early 1980s, so indicative fire histories could be compiled relatively cheaply for other areas.

Sources of error associated with satellite-derived fire history maps have been discussed by others (Allan *et al.* 2001; Bowman *et al.* 2003; Craig *et al.* 2002; Russell-Smith *et al.* 1997b) and also apply to this study. They include registration error (matching the image with the corresponding area on the Earths surface), cloud cover (limiting the coverage during certain periods of time), fire size (small fires are not detected, depending on image resolution), fire patchiness (unburnt areas within a fire scar may not be discernable), and fire scar fading (recovery of vegetation leading to discrepancies between the area mapped and area burnt). Apart from cloud cover, discussed previously, the main limitation of the quicklook method was that small fires were overlooked. These were either too small to detect on the satellite imagery (Yates & Russell-Smith 2002), or lacked associated hotspots, which precluded positive identification. Discounting small fires (under 20 ha) resulted in underestimation of the number of fires and area burnt, whereas the inability to discern within-fire patchiness tended to over-estimate the area burnt. Despite these constraints, the method is sufficiently robust for mapping fire history at the bioregional scale. The method is not suitable if a high level of precision is required, such as for point-specific analyses at the property scale. Its usefulness is restricted to broad assessment of the landscape, rather than individual fire scars, due to the inability to discern internal fire scar patchiness or small fires.

2.4.2 Regional Fire Patterns

The pattern of fire in the semi-arid Mount Isa Inlier had more similarity with arid Australian landscapes than mesic savannas. The area burnt each year was highly variable and strongly related to rainfall in terms of seasonal occurrence and area burnt. One good wet season (2001) triggered widespread fire; this is similar to central Australia where extensive wildfires occur following years of above average rainfall (Allan & Southgate 2002; Leigh & Noble 1981). This response reflects the effect of rainfall on fuel growth, as predicted by Walker (1981). I found a stronger correlation between the area burnt per annum and estimated pasture growth than for rainfall alone. The connection between antecedent rainfall and wildfire risk contrasts with temperate Australian forests where wildfire conditions are associated with drought (Leigh & Noble 1981), but supports findings for other Australian savanna areas (Russell-Smith *et al.* 2003)

There was less concurrence between the fire-season and fire-frequency results and Walker's 1981 model (Walker 1981). The Mount Isa Inlier falls within a 'fire season region' predicted by Walker to have a long fire season, beginning in late March and lasting until early January the following year. This estimate was based on fuel dynamics and considered the predicted response of different fuel types according to location, broad climatic zoning and expert opinion. The general pattern in the tropical savannas is that plant growth ceases after the wet season and flammability increases as fuel cures over the dry season.

The fire season detected by this study can best be interpreted by acknowledging the social component of landscape fire dynamics. For the Mount Isa Inlier, the fire season commences in August (approximately) and the area burnt increases dramatically over September and October, until the onset of summer storms that herald the beginning of the wet season. The area burnt declines as the wet season becomes established, but fires occur reliably through to March, more or less ceasing between April and July. This pattern reflects current pastoral land management in this region. Fires in the late dry season (August – November), either deliberately lit or accidental, spread easily in well-cured fuel. Fires are likely to burn until they reach roads, natural firebreaks or areas of insufficient fuel. With the onset of summer storms, fires are ignited by lightning or people. Pastoralists actively burn at this time to reduce 'woody weeds', to promote new growth in spinifex, which increases its palatability to cattle (P. McNeven, pers. comm., 2004), or to reduce fire hazard (R. Croft, pers. comm., 2004). Spinifex can burn when its moisture content is relatively high (Allan & Southgate 2002) and localised rainfall ensures fires will not spread extensively, so deliberate ignition continues over the wet season. Active burning ceases by the end of March in order to preserve remaining grass fodder for the coming dry season, and to prevent unwanted wildfire if there is no further rain. Unlike mesic savannas, where timing of fires is broadly categorized into 'early-dry season' or 'late-dry season' (Russell-Smith 2002a; Russell-Smith *et al.* 1997a; Williams *et al.* 2002), fires in the Mount Isa Inlier occur in the 'late-dry season' and during the 'wet season'.

The fire return interval in the Mount Isa Inlier was found to be 4-5 years, or longer, which is the latter end of the 1-5 year range predicted by Walker (1981). Fire frequency appeared to be driven by the regenerative capacity of spinifex (pers. obs.), as reported for other spinifex-dominated landscapes. For example, my values were comparable with the semi-arid Northern Tanami region, where 52% of the area remained unburnt over 6 years (Allan & Southgate 2002).

The results highlight the need to interpret fire patterns specific to an area if active land management is proposed. Generalisations made for broader areas could be misleading. The Mount Isa Inlier falls within Australia's tropical savannas where fires are generally reported to follow an early-dry season/late-dry season pattern, based on data from more mesic areas in the Northern Territory (Edwards *et al.* 2001; Russell-Smith 2002a; Russell-Smith *et al.* 1997b). The majority of mesic savanna fires occur in the late-dry season, with burning ceasing between December and March – which is the peak monsoon period when the country is usually too wet to burn. The customary fire return interval is 1-2 years in such mesic areas (Williams *et al.* 2002), unlike the 4-5 year interval found here. Also, there is no relationship between area burnt per annum and rainfall in the preceding year, since the grassy understorey (fuel) recovers every wet season (Felderhof & Gillieson 2006, Appendix 1). My results show that although the Mount Isa Inlier is located in the tropical savannas area, there is significant inter-regional variation. Fire patterns are strongly influenced by climatic factors at a more localised scale.

This suggests that calls to reduce the frequency of 'late-season, high intensity fires' across the tropical savannas (Russell-Smith 2002a; Russell-Smith *et al.* 2002; Williams *et al.* 2002; Woinarski *et al.* 2004) may be less urgent for the Mount Isa Inlier. The majority of fires (86.3%, n = 482) were less than 100 km^2 in size and 96.5% were less than 300 km^2. While extensive wildfires do occur, the largest fire recorded during the 1998 to 2003 study period was 1,500 km^2, considerably smaller than the 10,000 km^2 fires described for spinifex country in arid Australia (Allan & Southgate 2002). This may be an artefact of the period sampled, but according to local informants (Bob McDonald, pers. comm., 2003) the very large fires (1,500 km^2) experienced in 2001 were similar to extensive wildfires in 1975, which also followed a period of very high rainfall. The findings suggest that, by comparison, late season high intensity fires may be less of an issue in the Mount Isa Inlier. The trend to smaller fires in this area may have been the result of burning by pastoralists during late-dry season storms or during the wet season when fires were less likely to spread. Smaller fires could also reflect the rocky, broken country in the Mount Isa Inlier region, which is inherently different to many flatter savanna landscapes.

2.4.3 Implications for Research and Management

The above results can be used immediately to inform adaptive management. If protection of infrastructure or grass (cattle fodder) from wildfire is the prime concern, areas that have not burnt in four years or more are potentially hazardous. These areas could be assessed further to determine whether fuel reduction is warranted. Assessment should consider the fire history of the surrounding landscape. The spread of wildfire is minimised if the target area is surrounded by recently burnt vegetation. The strength of wet season rainfall should also be used as a guide, as time-since-fire is only an approximate indicator of wildfire risk. Alternatively, if the objective is to ensure that a range of fire ages is present, representation of post-fire successional stages can be determined from a fire history map. An area may require proactive burning to generate early fire recovery stages, or protection from fire to ensure some areas reach older stages. Areas burnt in the previous three years usually provide effective firebreaks, and small fires can be achieved by burning with storms at the onset of the wet season, or during the wet season itself.

From a research perspective, assessments of fire extent and frequency using remotely sensed data at the bioregional scale can assist researchers aiming to explain species decline or changes in vegetation structure in tropical savannas. Archived NOAA-AVHRR data is available back to the early 1980's and, although the scale may be relatively coarse, it contains an historical record of fire patterns (Craig *et al.* 2002). Such data could be coupled with GIS technology and interrogated with regard to land management practices, vegetation types, tenure, climate and other datasets (Craig *et al.* 2002) to help identify causes of change.

Returning to the patchiness paradigm (Chapter 1) and the suggestion that the goal for ecological fire management should be to implement fire regimes that develop and maintain fine-grained landscape patchiness (Russell-Smith 2002b), the results for the Mount Isa Inlier highlight two key issues. First, what is an appropriate degree of patchiness? Are fire-generated patches in the Mount Isa Inlier, which are typically under 100 km^2, 'small' enough? Second, current methods of mapping fire scars from satellite imagery over-estimate the area burnt, because they do not adequately record unburnt country within fire scars (Price *et al.* 2003; Yates & Russell-Smith 2002). The unburnt areas within fire scars contribute to fine-grained patchiness and are potentially significant for conserving species in fire prone environments (Ooi *et al.* 2006). Research is required to devise methods to measure patchiness at finer scales using satellite imagery, or at least to understand the variability in patchiness between and within fire scars. Research into the responses of fauna to different patch sizes in the burnt/unburnt mosaic is also required and would allow fire managers to have a more rationally defined goal than a vague 'fine-grained fire mosaic'. These issues are addressed in later chapters.

2.5 Conclusion

Fire patterns in the semi-arid Mount Isa Inlier had greater affinity with fire regimes reported for arid Australia than for mesic tropical savanna areas. In general, the study area is characterised by numerous small fires, spread over the late-dry season and the wet season, with a fire return interval of 4 to 5 years or longer. Seasonal rainfall and the fuel it produced were key drivers of fire extent and frequency; the area burnt each year was strongly related to rainfall in the preceding 12 months. The relatively small scale of burning had a significant affect on landscape-scale fire age. Approximately half of the study area did not burn over the six-year study period.

The dissimilarity between the semi-arid Mount Isa Inlier and more mesic savanna areas demonstrates that generalisations about current fire patterns in the tropical savannas, largely based on one region, need to be critically evaluated. The findings provide a context for planning fire management in northwest Queensland, but specific information is required on the responses of plant and animal species to fire if ecological fire management is to be undertaken.

3. VEGETATION DYNAMICS WITHIN FIRE SCARS

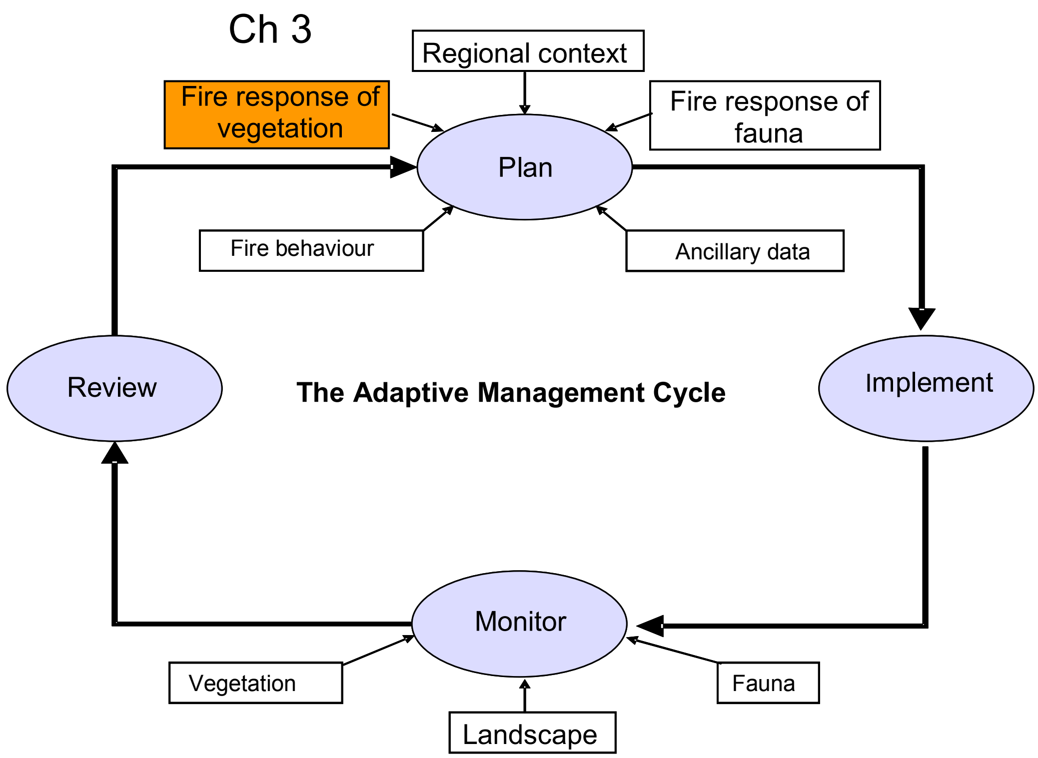

Figure 3.1 Information on the response of vegetation to fire is required so managers can define their desired result and evaluate the outcome of management actions.

3.1 Introduction

Determining the most appropriate fire regime for an area requires ecological insight (Anderson 2003). Knowing the system dynamics, particularly the responses of vegetation to fire, contributes to adaptive management in two ways. It is essential to monitoring and thus central to planning (Figure 3.1). Understanding the likely response of vegetation to fire enables managers to evaluate the outcome of particular management actions, which assists in defining desired results (Lindenmayer & Burgman 2005; Whelan 1995). It also contributes to monitoring effectiveness because indicators can be set that relate to species presence or absence, demography or extent of occurrence in the landscape, depending on the specific management goals. Monitoring then triggers management actions involving ongoing manipulation of the vegetation to achieve specific objectives. Finally, ground-level information on vegetation dynamics in response to fire is important for interpreting satellite imagery, which is being used increasingly for monitoring fires and recording fire history (e.g. http://www.firenorth.org.au).

Vegetation responds to fire at a number of levels. At the species level, fire can act directly by triggering seed release in serotinous species (Cowling *et al.* 1990; Gill 1997; Noble 1982), promoting germination by seed scarification (Auld & O'Connell 1991; Enright & Lamont 1989; Shea *et al.* 1979) or by smoke (Dixon *et al.* 1995; Enright *et al.* 1997; Roche *et al.* 1997; Williams *et al.* 2005). It can provide a cue to flowering (Bond & van Wilgen 1996; Bradstock & Auld 1995; McFarland 1990) or provide conditions that favour seedling establishment (Morgan 1998; Williams *et al.* 1994). After fire, some plants can sprout from epicormic buds, lignotubers, bulbs, corms or rhizomes, while others tolerate fire via thick bark or plant architecture that protects sensitive tissues (Keith 1996; Whelan 1995). These responses collectively determine the species assemblage in the vegetation community.

Community level changes over time depend on maturation times of different species and longevity. Changes in species composition, species dominance and vegetation structure have been recorded for different vegetation types (Gill 1999), but site-specific and fire-specific factors make it difficult for land managers to accurately predict fire responses (Whelan *et al.* 2002). To help forecast likely changes in the species assemblage, life history attributes have been grouped into general response categories (functional groups) (Bond & van Wilgen 1996; Gill *et al.* 1981; Lindenmayer & Burgman 2005; Whelan 1995). Common groupings relate to plant persistence after fire (sprouters and non-sprouters) and plant recruitment after fire (fire-stimulated or not) (Bond & van Wilgen 1996). The recurrence of fire over time is a critical consideration, however (Andersen 2003; Gill 1975; Keith 1996; Whelan 1995; Whelan *et al.* 2002), and extrapolating from the outcome of a single fire event requires caution. A better understanding of species composition and community structure can be obtained by applying models that link fire regimes with functional groups to explain post-fire ecosystems (Noble & Slatyer 1978; Cattelino *et al.* 1979, Lavorel & Garnier 2002). To aid prediction, these models use plant adaptive traits, their likely persistence and establishment under different conditions, and the periodicity of disturbance.

Models using functional groups only provide an approximate guide to post-fire outcomes because plants respond differently depending on fire intensity (Morrison & Renwick 2000), which is highly variable within a given fire (Atkins & Hobbs 1995; Tolhurst 1995; Whelan 1995). Fire intensity affects plants by influencing exposure temperatures (Williams *et al.* 2003a), although total heat output, duration of heating and the moisture conditions of the soil and vegetation also affect the ecological response (Tolhurst 1995). A confounding factor is that some species exhibit different regeneration strategies in different environments. For example, several species of spinifex resprout in some localities but act as obligate seeders in others (Rice & Westoby 1999).

The aim of this study is to provide information on responses of vegetation to fire in the Mount Isa Inlier. An efficient approach is required to gain an overall understanding for ecological fire management because there is only one published study on plant/fire interactions in this region, and it relates to a single fire event (Williams *et al.* 2002). The approach taken here integrates key components that drive vegetation/fire dynamics (Figure 3.2). Emphasis is on general patterns at the landscape scale, concentrating on dominant species, post-fire responses and variability. Results from the semi-arid Mount Isa Inlier are compared with vegetation responses documented for more arid spinifex landscapes and more mesic savannas.

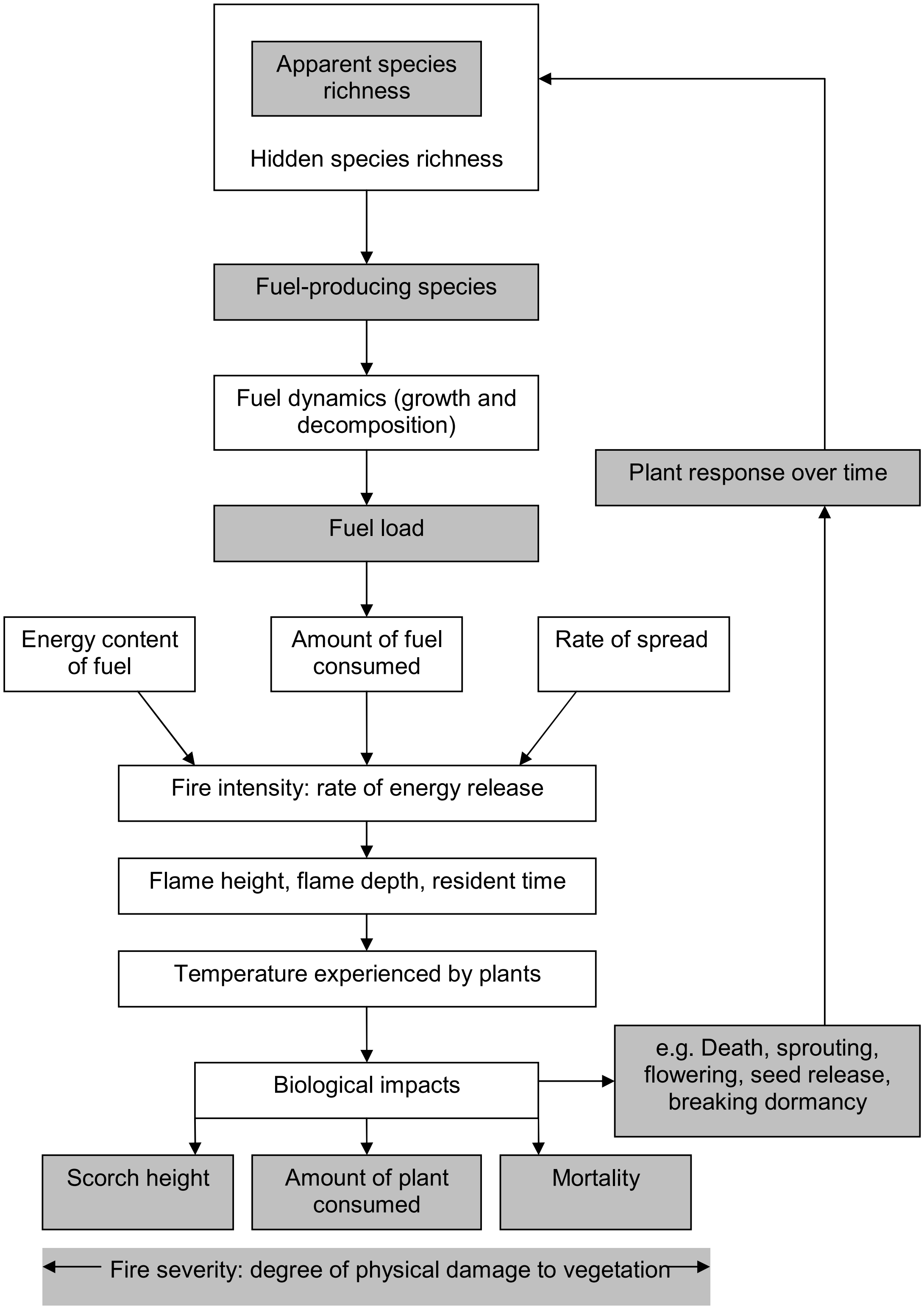

Figure 3.2 A generalised diagram showing the relationship between fire and vegetation. Aspects covered in this chapter are shaded grey.

Species composition and recovery over time is assessed by measuring changes at sites of different fire-age (time since fire). Information on maturation time is recorded so suitable fire free intervals can be provided for seed production, seedling establishment, and subsequent plant growth and reproduction (Andersen 2003). Species are assigned to functional groups to facilitate comparison with other areas.

Plant species richness and the responses of individual species are affected by the fire regime, which in turn is dictated by the fuel load (Gill 1999). Fire frequency, for example, depends on the time to accumulation of sufficient fuel to sustain a fire; and fire intensity is highly dependent on the type and amount of fuel. The importance of the fuel load is best explained by revisiting Byram's (1959) fire intensity equation: **I = HwR** [where I = Fire intensity, H = heat yield per unit area, w = the amount of fuel consumed and R = rate of forward spread]. The heat yield is related to the type of fuel, which will be relatively constant for a given vegetation type. The quantity of available fuel, 'w', has a disproportionate affect on intensity. Generally, as the fuel weight doubles, the rate of forward spread also doubles (Luke & McArthur 1978), increasing fire intensity four-fold. This relationship is indirect, however, and takes effect where changes in fuel load reflect changes in fuel continuity (Cheney & Sullivan 1997). Weather, terrain and fuel factors in combination determine fire behaviour; but fuel load can be practically modified and its manipulation is important for active fire management.

Most fuel in savanna landscapes comes from the dominant grasses (Gill 1999), which are usually *Triodia* spp. (spinifex) in the Mount Isa Inlier. In grassy woodlands with a regular understorey structure, fuel depth can be used to estimate fuel load (Bransby & Tainton 1977). However, this method has not been adopted in the spinifex hummock grasslands of the Mount Isa Inlier. This study calibrates fuel-depth against fuel-load for spinifex communities, thus providing an inexpensive, rapid method for estimating fuel load in this region where spinifex communities form a discontinuous fuel type.

Fire intensity and associated heat output are impractical to measure in broad scale studies, so a series of indicators are used to estimate fire severity. Studies have confirmed a direct link between measured fire intensity and post-fire indicators of intensity (Williams *et al.* 2003b). These include leaf scorch height, leaf char height, proportion of grass layer burnt and shrub twig diameters (Moore *et al.* 1995; Williams *et al.* 2003b). This study uses char and scorch heights, unburnt patches of ground and shrub twig diameters to indicate differences in fire intensity between and within fire scars. Variation in fire intensity within fire scars has been widely acknowledged (Tolhurst 1995; Turner *et al.* 2003; Whelan 1995), but rarely quantified. The variability is such that different fire treatments can be experienced by plants in close proximity (within metres) even in a reasonably intense fire (Atkins & Hobbs 1995). This heterogeneity has implications for the long term persistence of biota (Atkins & Hobbs 1995; Lindenmayer & Burgman 2005; Ooi *et al.* 2006; Whelan 1995) because it is the fire temperature in localized patches that gives rise to ecological effects. Spatial heterogeneity is obscured in most studies as temperatures are averaged over large areas when reporting fire intensity (Whelan 1995).

Understanding the inherent variability of fire is important for developing the patchiness paradigm and for planning the use of fire to manage vegetation. Spatial variation or patchiness within fire scars will make the next fire more variable, leading to a cyclic pattern of heterogeneity. However, emphasis on internal fire patchiness may not be necessary, allowing managers to focus on a defined fire size as their management goal. This chapter investigates post-fire responses in spinifex/snappy gum communities. The aim is to gain an overall understanding of landscape dynamics, allowing the effects of different fire regimes to be anticipated.

3.2 Methods

3.2.1 Vegetation Recovery After Fire

Site stratification and sampling

Vegetation was sampled at 75 sites across the bioregion, representing 5 different times since fire (<1 year, 1-2 years, 2-3 years, 3-4 years and 4 years or more) (Figure 3.3). Sites were identified using the fire history map presented in Chapter 2. A geographic information system (GIS) was used to identify areas of suitable vegetation (spinifex/snappy gum communities), with reasonable access (more than 100 m but less than 1 km from an access track), and spread of fire ages. Where possible, sites were located at least 1 km from the fire scar edge. Boundary and grazing effects were documented in field notes.

A random point generator was used to specifically locate sample sites, except for fires less than one year old that had not been mapped at the time of survey. Satellite-generated hot-spot data were used to locate these sites. The intention was for sites of each fire age to be spread equally between north and south. This was precluded by the small percentage of the landscape burnt in some years and access difficulties. Some site locations were adjusted in the field as access tracks no longer existed, reflecting the inadequacy of the digital road data. In these instances, sites were sampled where the best access to a fire scar could be obtained.

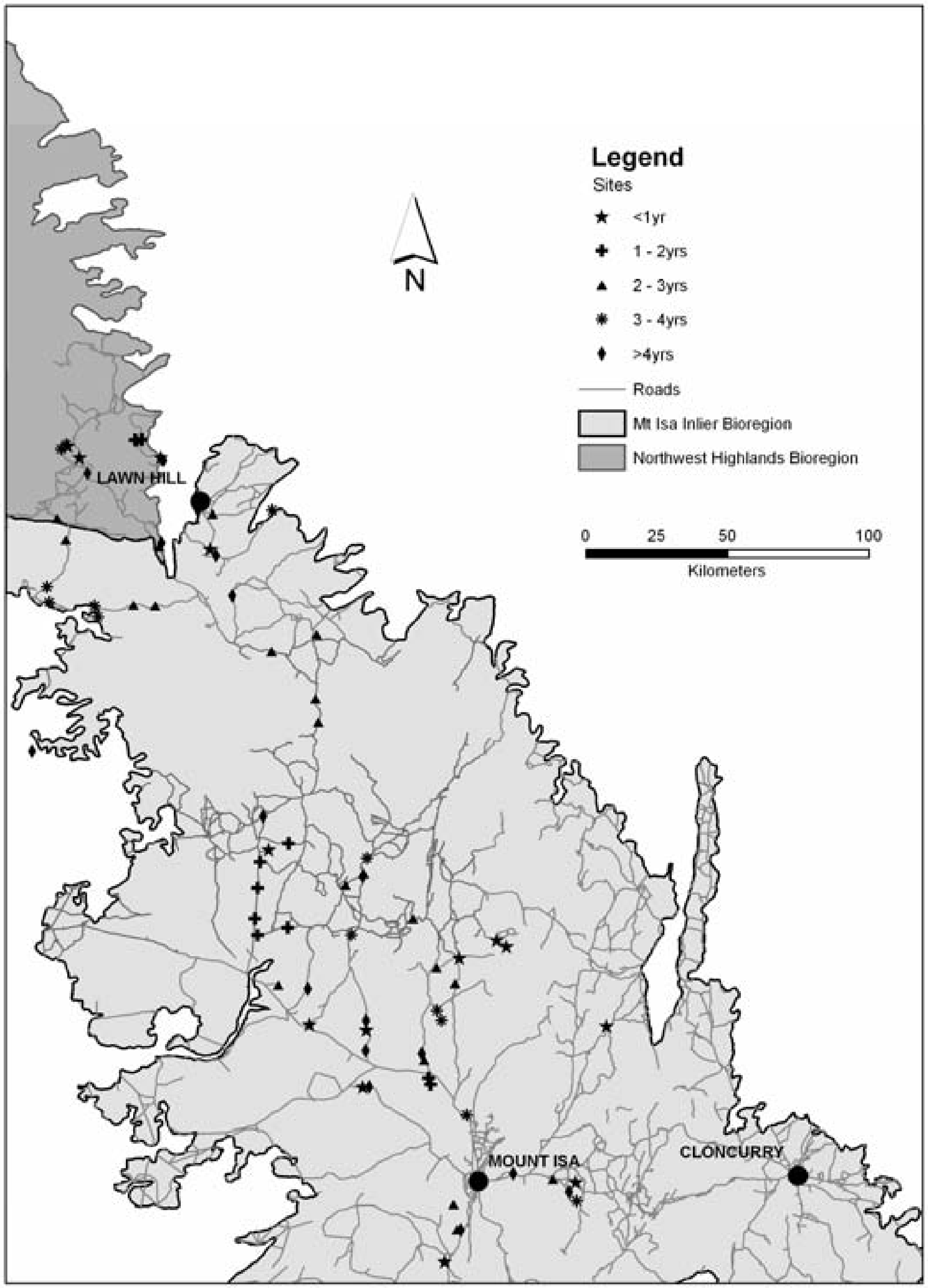

Figure 3.3 Vegetation was sampled at 75 sites across the bioregion. Sites were stratified with respect to 'time since fire'; five fire-age classes were represented (see text).

The wheel-point method (Griffin 1989) was used to measure and analyse vegetation pattern and post-fire recovery trends. The wheel-point is a modification of the point-cover method and enables plant community composition to be recorded by cover and vertical structure. A spoked, rimless wheel is pushed across the landscape and a 'hit' is recorded when a marked spoke intercepts cover on or above the ground (Figure 3.4). Data were recorded at this point as well as vertically above the point if canopies of shrubs or trees were present. The method allows for rapid assessment over large areas and was chosen so relative comparisons could be made between sites. The level of accuracy and repeatability were considered acceptable given that emphasis here was on the fire responses of the dominant species.

A hand-held GPS (Garmin GPSMAP 76) was used to navigate to each site. Data were collected on general site characteristics such as dominant vegetation, slope, aspect, lithology, landscape position, disturbance by grazing and signs of fire. An index of herbivory was considered, but accepted measures such as recording the density of cattle pads (Landsberg *et al.* 1999) was not appropriate in this geologically complex environment. The most heavily grazed site had been devastated by grasshoppers, not cattle, and macropod droppings were more common than cow pats. Cattle were only one component of the total grazing pressure, suggesting that that the grazing indices used in productive landscapes may need to be modified for use in more natural landscapes.

Transects were orientated away from access tracks and aligned across the slope to minimise the effects of changed fire intensity as fire proceeded uphill (Cheney 1981), or enhanced soil fertility or soil moisture down-slope. Initial transect direction was recorded, although direction changed occasionally in order to remain within the fire scar or to maintain position on a slope. The location of each directional bearing change and the new bearing were recorded. Location data were also collected at the middle and end of each transect.

Data acquisition required two people, one to operate the wheel-point and the other to record 'hit' data. The same person recorded 'hit' data each time to ensure consistency in determining what constituted a strike upon a plant. Each transect was 400 m long, providing data at 200 points. The adequacy of this sampling intensity was established by calculating running means after initially recording data along 600 m transects.

Figure 3.4 The wheel-point apparatus was used to sample plants in the ground, shrub and tree layers.

At each point, data were recorded on ground cover, and canopy cover of shrubs and trees. If plant species were not known, voucher specimens were collected for later identification at the herbarium. Plants were classified according to: height (in 25 cm increments); maturity (seedling, juvenile, adult or old);

reproductive status (budding, flowering, fruiting or seeding); and regenerative traits (sprouting from the base, trunk or both, sprouting apically, or dead).

Voucher specimens were identified upon return to the laboratory. Fertile material was not always available, which prevented identification to species level. In these cases, the plant was identified to genus and given a unique numeric identifier (e.g. *Acacia* 1) based on morphological differences. A comprehensive species list was compiled and literature references and field observations were used to categorise the species according to their functional groups (annuals, ephemerals or perennials; obligate seeders or resprouters).

Data analysis

An hierarchical approach was used to analyse data. First, patterns of species richness and dominance were considered at the vegetation community level. Second, within the community level, data were investigated to classify species into structural and functional groups. Third, within these groups, dominant species were selected to illustrate information requirements for planning fire to conserve particular species.

At the vegetation community level, the cover of each species was determined by calculating the percentage of hits on that species out of the total number of transect points. The sum of percentage covers of each species gave a cover value for the whole site and for ground cover, shrub canopy cover and tree canopy cover. Analysis of variance and Spearman's Rank correlation coefficients were used to determine whether the number of species present and percentage cover varied significantly ($p < 0.05$ level) with fire age.

Simpson's Diversity Index (Ludwig & Reynolds 1988) was used to investigate the patterns of species and structural diversity with time since fire. For species diversity, the index was assigned to each site based on the number and dominance of species present. Height diversity was determined using the same equation, but height classes were substituted for species. Thus, Simpson's Index provided a single value that integrated the number of height classes present and the number of occurrences of each height class. These indices were calculated for an entire site and then calculated separately for the ground cover, and the shrub and tree canopy cover components. The relationships between species and height diversities with time since fire were investigated using analyses of variance and Spearman's Rank correlations.

Gill (1999) promoted rank-cover curves for showing broad vegetation changes after fire, as these graphs allow for the relative performance of species to be tracked over time. Plotting Log_{10} of cover against species rank (in terms of dominance) over time allows changes in site richness and dominance to be clearly shown. To produce rank-cover curves, a list of the species recorded at each fire age was compiled, along with the total cover for that species across all sites of similar fire age. Species were ranked numerically by cover, and rank-cover curves were produced for each fire age at the site level and for each stratum.

Species were assigned into groups to understand the dynamics of change, based on the approach described by Gill (1999). Four classification options were used to aid interpretation: plant maturity, reproductive status, regenerative traits and longevity (Table 3.1). The entire data set (75 sites) was used for calculations related to maturity, reproduction and regeneration. At each site, and for each stratum, data were collated by finding the percentage of occurrences in each category. For example, for plant maturity, the proportion of ground cover species that were seedlings, juveniles, adult or old aged (showing signs of senescence) was determined. Similarly, reproductive status was summarised according the proportion of plants that were budding, flowering, fruiting or seeding. Spearman's Rank correlation coefficients were used to investigate changes in occurrence over time, with analyses undertaken separately for each stratum. In some cases categories were grouped to increase sample size, such as ‘seedlings and juveniles’, ‘adults and old’, or ‘reproductive’, ‘not reproductive’. This also helped to overcome issues related to the season of reproduction, as data collection was restricted to a single sampling season.

Table 3.1 Data on plant maturity, reproductive status, regenerative traits and longevity were recorded for every plant intercepted by the wheel-point apparatus. In later analysis, some categories were grouped (grey shading).

Plant maturity		Reproductive status		Regenerative traits	Longevity
Seedling	Seedling & juvenile	Budding	Reproductive	Resprouting from base	Annual
Juvenile		Flowering		Resprouting from trunk	Ephemeral
Adult (reproductive age)	Adult & old	Fruiting		Apical growth (e.g. grass tree)	Perennial
Old aged (signs of senescence)		Seeds dispersed		Dead	
		No reproductive material	Not repro.		

Pooled data were used for analyses of functional groups based on longevity and mode of regeneration. For each fire age, plant species were categorised as ‘short lived’ (annuals and ephemerals) or perennials, and ‘seeders’ (can only regenerate from seed) or ‘sprouters’ (can regenerate via subterranean or aerial buds). The proportion of species with these responses, and total cover in each response category, were summarised for each fire age.

Analysis is required at a finer scale to address the needs of particular species. For example, information on maturation time enables suitable fire intervals to be determined because adequate time is required between fires to allow plants to grow and reproduce. To assess the fire interval requirements of the dominant species, data on maturation and reproduction were extracted for *Triodia* spp., *Acacia chisholmii* and *Eucalyptus leucophloia* where these were the first or second dominant species of the site's ground, and shrub or tree canopy layers respectively. The relationship between the species' percent occurrence in each maturation and reproductive category, and time since fire was investigated using Spearman's Rank correlations.

3.2.2 Fuel Load Accumulation Over Time

Fuel load accumulation over time was assessed at the same 75 sites where the vegetation response data were collected. A piece of flagging tape weighted with a metal washer was dropped every 10 m while traversing each 400 m transect. Measurements of fuel depth were recorded at each of these sample points as this provided a rapid, inexpensive surrogate for estimating fuel load. Subsequent calibration was based on quadrat-based harvesting (see below: calibrating the 'disc-dropper'), allowing fuel load (t/ha) to be predicted from fuel depth.

Fuel depth was measured using a 'disc-dropper'. This consisted of a plastic disc, 35 cm in diameter and weighing 241 g, inserted over a 2 m wooden rod marked off in cm. The rod was put vertically into the fuel bed, ensuring that its base rested on the soil. The disc was dropped from a standard height of 50 cm above the fuel and its resting height was read from the rod. For consistency, fuel height was measured where the plastic disc was in contact with the rod, as the disc sometimes rested at an angle, especially if it hit the side of a spinifex hummock. The same disc was used for the entire sampling period to keep its weight constant. 'Running means' calculated after sampling the first three transects showed that 40 points were adequate for estimating fuel load.

Calibrating the 'disc-dropper'

Fuel depth measurements at each site were converted to t/ha to enable comparisons to be made with other savanna areas. The disc-dropper was calibrated for two groups of spinifex: hard (*Triodia longiceps* and *T. molesta*) and soft (*Trioda pungens*). Hard spinifex species were grouped due to difficulties in identifying spinifex plants to species level in the field when fertile material was not available. Grouping species into two categories was considered valid as more than one type of spinifex was present at many sites and their fuel characteristics are similar.

Calibration sites were based on areas previously visited to record vegetation. Emphasis was on sites recently burnt (< 3 years), 3 - 4 years post fire, or more than 4 years post fire to ensure a wide range of fuel depths. Hard spinifex was sampled in eight different areas, each with two transects approximately 20 m apart. Three to four quadrats were sampled on each transect, depending on time constraints. Soft spinifex was sampled from seven different areas, again along two parallel transects, but sampling four quadrats each time. Fuel was harvested from 106 quadrats in all.

Calibration involved laying out a 20 m measuring tape and taking samples from a 1.5 m by 0.5 m quadrat that was placed adjacent to the tape every 5 m. Three fuel depth measurements were taken, one from each third of the quadrat (Figure 3.5) and the number of spinifex hummocks, their length, breadth and proportions in the quadrat were noted. All live and dead ground layer material was removed, cutting at ground level using a cane knife or secateurs as necessary. Fuel was spread onto a plastic sheet and separated into spinifex, other grass and fine dicotyledons, and coarse material (over 6 mm in diameter). These samples were placed in paper bags that had been labelled and weighed prior to the field trip. Each sample was weighed in the field using a hanging scale and sample details were recorded on the label. A sub-sample was taken when large quantities of fuel were present, in which case, the total fuel weight and the sub-sample weight were noted. Upon return to the laboratory, samples were dried at 80^0C for 48 h and reweighed to obtain the dry weight of fuel. The average fuel depth was then calibrated against the dry weight of fuel in each quadrat.

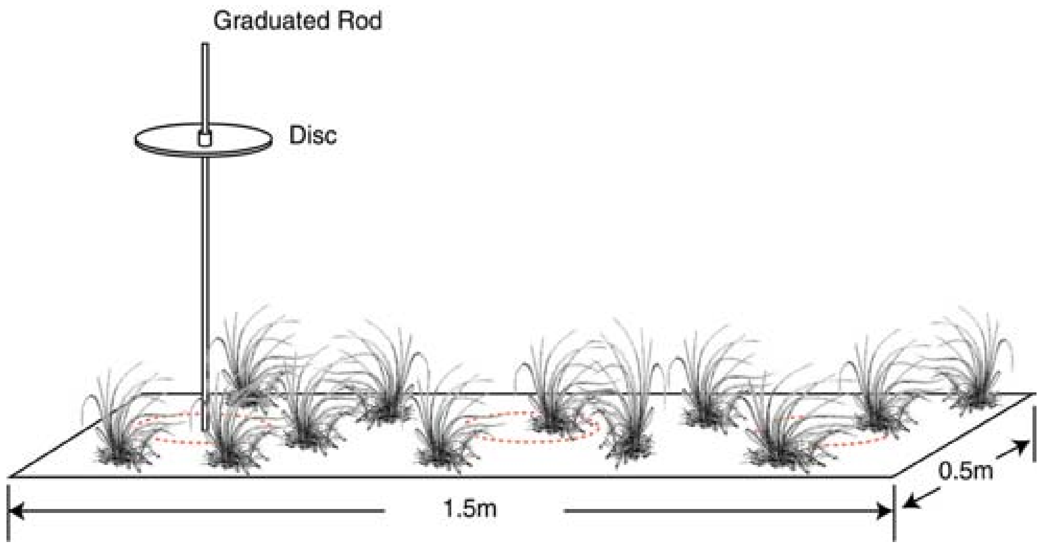

Figure 3.5 The 'disc-dropper' was calibrated by relating the average depth of fuel (based on 3 measures) to the dry weight of fuel collected within a 1.5 m x 0.5 m quadrat.

Relationships were explored between fuel height, fuel weight, fuel type (spinifex, fine dicotyledons or coarse material), areal cover of spinifex, and hummock volume, to confirm that disc drop height was suitable for estimating fuel load. The resultant regression equation was used to provide an estimate of fuel load (t/ha) for the 75 sites across five fire-age classes. These data were used to investigate fuel load accumulation over time.

3.2.3 Variation in Fire Severity

Variation in fire severity between fire scars

To assess variation in fire severity (as an indicator of intensity) between fires, indicators of fire severity were recorded along transects at 19 areas that had burnt between October 2004 and March 2005 (Figure 3.6). Potential sites were identified using NOAA and MODIS satellite data. Both satellites provide hotspot and fire scar information, but with different sensitivities and at different resolutions. Some field sites were based on hotspot data alone, with no fire scar, while others had a fire scar but no hotspots. Map co-ordinates for each site were determined prior to fieldwork using a GIS to identify burnt areas close to roads or access tracks. Where the pre-determined points proved inaccessible, transect start points were chosen where access tracks intersected with a burnt area. Start points were located well within the burnt area and at least 50 m away from the access track to minimise the effects of disturbance. Each site had low open eucalypt woodland with a spinifex understorey; however, *E. leucophloia* (snappy gum) was not always the dominant tree species because site locations were dictated by recent fire occurrence.

Two pairs of field workers estimated fire severity at each site. Data were recorded along two transects, which were roughly parallel and orientated perpendicular to the access track. This helped to ensure that core areas of fire scars were sampled, in case fires had been extinguished by back-burning from roads. Occasional sites had very rugged terrain, in which case transects were orientated to ensure that both pairs of workers sampled similar vegetation and landform. The starting point of the first transect was located as described above; the second transect started approximately 25 to 50 m south (determined by metre-length pacing). The wandering quarter method (Figure 3.7) (Mueller-Dombois & Ellenberg 1974) was used to collect data. This involved laying out a 90^0 angle at the starting point, and bisecting it with an imaginary line based on the compass bearing perpendicular to the track. The closest tree in the 90^0 quarter was measured and became the point for a second 90^0 quarter laid out in the same compass direction. Again, the closest tree in the quarter was measured, with the process continuing until 25 trees were sampled.

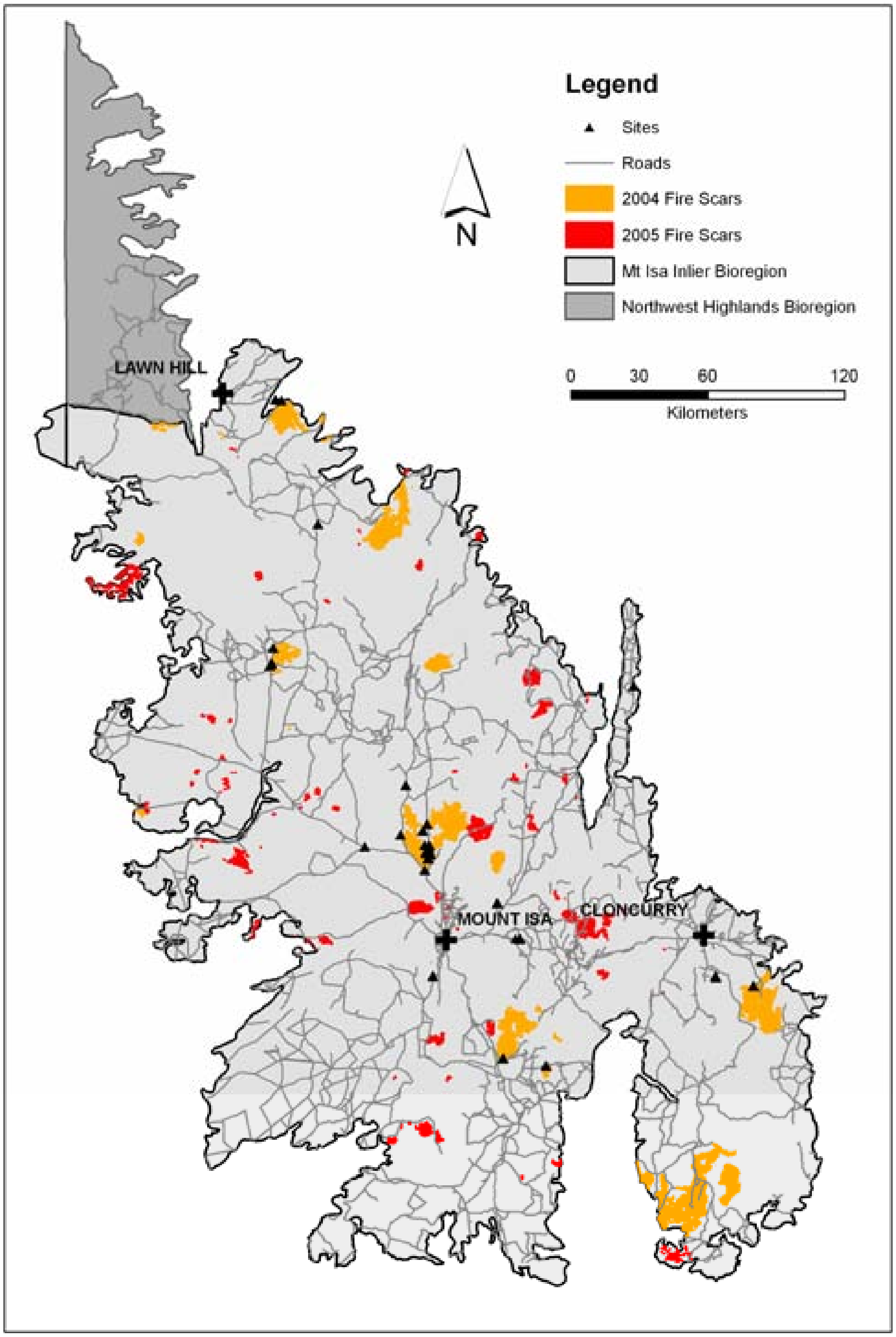

Figure 3.6 Sites (black triangles) where indicators of fire severity were recorded. Sites were located in areas that had burnt within the previous 6 months, although not all fire scars were mapped.

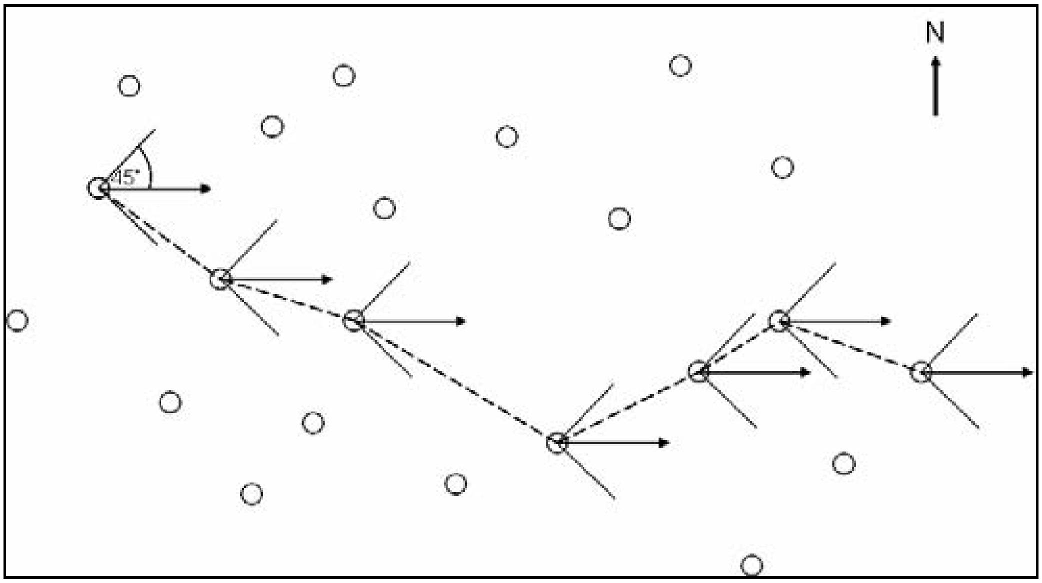

Figure 3.7 The wandering quarter method for plotless sampling of vegetation (Mueller-Dombois & Ellenberg 1974). Open circles represent trees.

The tree species was noted at each point and map co-ordinates were recorded using a hand held GPS (Garmin GPSMAP 76). The height of char (burnt bark) on the trunk or branches was measured using a 2 m staff marked off in cm. The proportion of the tree's canopy removed by fire was visually estimated and assigned to one of five categories (no scorch, <1/3 scorch, 1/3 - 2/3 scorch, >2/3 scorch, total canopy scorch).

To sample the shrub layer, measurements were made on the first shrub encountered enroute to the next tree. The shrub species was determined from bark characteristics, growth form, desiccated leaves in situ, resprouting leaves if present and inference based on nearby unburnt specimens. Shrub and char heights were recorded and electronic callipers (Fragram ® Digital Vernier Calliper, C342) were used to record the diameter of up to 5 burnt twig-ends. Measurements were made as close to the burnt end as possible, usually about 0.5 cm from the distal point. If more than 5 twigs were available for measurement, the most central twig and one from each of the north, south, east and west quadrants were measured. Up to twenty-five shrubs were measured on each transect, depending on the number of shrubs encountered between points.

The number and length of unburnt spinifex patches was recorded between each point to give an indication of on-ground patchiness. Patch length was estimated by pacing in 1-m steps, with the same person in each pair providing the measurements each time. General vegetation, landscape, and fire scar descriptions were also recorded.

Variation in fire severity within a single fire scar

Indicators of fire severity were recorded at 10 sites within one fire scar to assess internal variation in fire severity. The methods were the same as for between fire scar comparisons.

Data analysis

Data for each transect were summarised for:

- Number of tree and shrub species present;
- Number of shrubs measured;
- Mean, standard deviation and standard error of tree and shrub char height;
- Mean, standard deviation and standard error of tree canopy scorch;
- Mean, standard deviation and standard error of shrub percentage charred (this was considered a better indicator than shrub char height due to height differences between different shrub species);
- Mean twig diameter per shrub and the mean, standard deviation and standard error of all twig diameters measured on the transect;
- The percentage of twigs in 5 different size classes (>1 mm, >2 mm, >2.5 mm, >3 mm, and >4 mm); and

- Transect length, the number of unburnt patches per m, and average length of unburnt patches.

The data were further summarised by assigning scores for severity and patchiness to each transect. These scores were derived as follows. First, the key variables to indicate fire severity were considered to be tree char height, leaf scorch (canopy removal), proportion of shrub charred, the diameters of twigs left on shrubs and the number of unburnt patches per metre. Data from all 56 transects were used to identify the maximum and minimum values for each of these variables. From the spread of values, severity classes were determined ranging from 1 (indicative of low severity) to 5 (indicative of high severity). For example, using these variables, a severe fire would typically have complete canopy scorch, high char heights, shrubs entirely charred, thick twigs and no unburnt patches. Data for each transect were assessed against these criteria, and the median value for the five variables gave the final transect fire severity score (Table 3.2)

Site patchiness scores were determined in a similar fashion (Table 3.3); however, standard errors were used rather than means as they provided a better indicator of variability. Large standard errors for tree and shrub char height, canopy scorch, percentage of shrub charred and twig diameters indicated greater fire heterogeneity than low standard errors. The number and length of unburnt patches was also considered, with large patches and a large number of patches per metre giving a higher patchiness score. Again, the data range for each variable was examined before dividing the values into 5 classes representing different levels of patchiness from 1 (low) to 5 (high). The median of the seven scores provided the overall transect patchiness score.

Table 3.2 Fire severity scores were determined for each transect by assessing transect values against severity classes assigned to 5 different variables. The final severity score was the median of the five scores.

Variable	Severity class 1	2	3	4	5	Transect value (example)	Score
Mean char ht (cm)	0-100	101-200	201-300	301-400	>400	303.4	4
Mean canopy scorch (%)	0	0.1-33	33.1-66	66.1-99	100	75.1	4
Mean % shrub charring	0	0.1-25	25.1-50	50.1-75	75.1-100	52.5	4
# unburnt patches/m	>0.06	0.041-0.06	0.021-0.04	0.001-0.02	0	0	5
Twig diameter distribution (% per size class)	37.6	52	4	3.2	3.2		5
Criteria for twig severity classes	*>20%, none in 4 or 5*	*<20% in 1, none in 4 or 5*	*<20% in 1, records in 4 or 5*	*<20% in 1, <3% above 4mm*	*If >3% above 4mm*	**MEDIAN**	**4**

Table3.3 Fire patchiness scores were determined for each transect by assessing transect values against classes assigned to 7 different variables. The final patchiness score was the median of the seven scores.

Variable	Patchiness class 1	2	3	4	5	Transect value (example)	Score
SE tree char ht (cm)	0-12	12.1-24	24.1-36	36.1-48	>48	31.3	3
SE canopy scorch (%)	0	0.1-2	2.1-4	4.1-6	>6	4.8	4
SE shrub char ht	0	0.1-20	21-40	41-60	>60	12	2
SE % shrub charred	0-5	5.1-10	10.1-15	15.1-20	>20	6.1	2
SE twig diameter (mm)	0-0.5	0.51-1	1.01-1.5	1.51-2	>2	0.08	1
# unburnt patches/m	0	0.01-0.02	0.021-0.04	0.041-0.06	>0.06	0	1
Mean patch length (m)	0-2	2.1-4	4.1-6	6.1-8	>8	n/a	

Point-based fire severity estimates were also made using an index to integrate data at each point. The severity index was determined as above, after considering the data range for all 1392 points. Each indicator value was converted to a 1-5 scale and the median of scores for char height, canopy scorch, percentage shrub charred and mean twig diameter provided the point-based severity score. The number of unburnt patches per metres was not relevant at this scale.

Data were checked for normality and transformed (Log_{10}) if this improved their statistical distribution. Some variables could not be easily or realistically transformed, so some analyses were restricted to non-parametric tests. Spearman's Rank correlation coefficients were calculated to explore relationships between canopy scorch, char height, shrub charring, twig diameters and the number of unburnt patches per metres. As these are established indicators of fire severity (Williams *et al.* 2003b), strong relationships were expected. Patchiness was expected to decrease as fire severity increased, so relationships between patchiness and severity indices were also investigated. Data were analysed at transect, site and fire scar levels.

Preliminary analysis showed that the fire severity variables were collinear, so the two that best conformed to normality were used for subsequent parametric analyses. These were Log_{10} tree char height and Log_{10} twig diameter. One-way analyses of variance were undertaken to test for significant differences in the means of these two variables with respect to the fire scar number, site, tree or shrub species, and month of burning. Analyses of variance were also used to test for differences between sites within a single fire scar. A paired t-test was used to compare the means for tree char height between transect pairs. Between-transect tests were not carried out on shrub data since shrubs were not present at all sites, so sample sizes were unequal.

The fire severity scores provided a final test for differences in fire severity between and within fire scars. An overall severity score, rather than a specific measure, allowed for standardisation and simplified comparisons between sites with different tree species, shrub species and vegetation structure. The non-parametric Kruskal-Wallis test was used to compare differences in severity scores between fire scars, sites and for sites within a single fire scar. The Mann-Whitney U test was used to test for differences in fire severity scores between transect pairs. Kruskal-Wallis tests were also used to compare the number of unburnt patches between fire scars and at fire scar, site and transect levels.

3.3 Results

3.3.1 Vegetation Recovery After Fire

Species diversity and dominance

Spinifex was the dominant ground cover at all sites, ranging from 56.8% to 91.2% of all ground cover hits. In other savanna areas, buffel grass (*Cenchrus cilciata*) and other invasive grass species are known to significantly alter fuel configurations and fire characteristics (Butler & Fairfax 2003; Fairfax and Fensham 2000), but this was not apparent in the area studied. Buffel grass was evident along some roadsides but only recorded incidentally at any of the sites (0% to 1.1% of all ground cover hits). *Acacia chisholmii* was the most common shrub species, and *Eucalyptus leucophloia* was the dominant tree species. The species recorded at each fire age are listed in Table 3.4. The number of species per site did not vary with fire age.

Cover data were transformed (Log_{10}) and analyses of variance showed that the means of Log_{10} % cover were significantly different between different fire ages for ground cover ($F = 53.12$, $p < 0.001$, $df = 4$), tree cover ($F = 3.71$, $p < 0.01$, $df = 4$) and total cover ($F = 44.63$, $p < 0.001$, $df = 4$), but not Log_{10} % shrub cover. Spearman's Rank correlation coefficients showed that cover increased with time since fire for ground cover (R = 0.851, $p < 0.001$, n = 75), tree cover (R = 0.324, $p < 0.01$, n = 75), and total cover (R = 0.856, $p < 0.001$, n = 75).

Table 3.4 Plant species recorded at sites of different fire age. Lists are from pooled data for 75 sites across the Mount Isa Inlier bioregion. Asterisks show species that accounted for 2% or more of the intercept points for each fire age.

Fire Age Class				
< 1 year post-fire	1 - 2 years post-fire	2 - 3 years post-fire	3 - 4 years post-fire	> 4 years post-fire
Ground cover				
Bulbostylis sp.1	*Alternanthera nodiflora*	*Aristida sp.*	*Aristida sp.*	*Aristida sp.*
Cenchrus ciliaris	*Aristida sp.3**	*Aristida sp.2*	*Aristida sp.1*	*Boerhavia pubescens*
Crotalaria sp.	*Blumea sp.1*	*Astrebla sp.1*	*Boerhavia paludosa*	*Cenchrus ciliaris*
Cyperus sp.	*Cenchrus ciliaris*	*Astrebla sp.2*	*Chrysopogon fallax*	*Cymbopogon bombycinus*
Dodonaea lanceolata (?)	*Chamaesyce sp.**	*Boerhavia pubescens*	*Dichanthium sp.*	*Dichanthium sp.1*
Enneapogon polyphyllus	*Cyperus sp. (?)*	*Chrysopogon fallax*	*Enneapogon polyphyllus*	*Digitaria sp.*
Eriachne ciliata	*Dichanthium sp.*	*Cymbopogon bombycinus*	*Eriachne sp.3*	*Enneapogon polyphyllus*
Fimbristylus sp.1	*Dichanthium sp.2 (?)*	*Cyperus sp.*	hard spinifex*	*Eriachne ciliata*
Fimbristylus sp.2	*Digitaria sp.*	*Dichanthium sp.*	*Heteropogon contortus*	*Eriachne sp.1*
Goodeniaceae sp.1	*Enneapogon polyphyllus*	*Dichanthium sp.1*	*Iseilema sp.1*	*Fimbristylus sp.*
hard spinifex*	*Enneapogon robustissimus*	*Dichanthium sp.3*	*Mnesithea formosa*	*Glycine sp.*
Indigofera colutea	*Fimbristylus sp.*	*Digitaria sp.1*	*Panicum decompositum*	*Goodeniaceae sp.1*
Indigofera linifolia	*Galactia sp.1*	*Enneapogon polyphyllus*	*Schizachrium fragile*	hard spinifex*
*Indigofera sp.**	*Goodenia armitiana*	*Enneapogon sp.1*	*Solanum quadriloculatum*	*Heteropogon contortus*
Ipomoea sp.	*Goodeniaceae sp.1*	*Eriachne sp.1*	*Sporobolus australasicus*	*Indigofera sp.*
*Mnesithea formosa**	hard spinifex*	*Fabaceae sp.1*	*Themeda australis*	*Iseilema sp.1*
Other forb sp.1*	*Indigofera sp.*	hard spinifex*	*Tribulopsis sp.1*	*Ishaemum australis*
Other grass'	*Indigofera sp.1**	*Heteropogon contortus*	*Triodia pungens**	*Mnesithea formosa*
Panicum decompositum	*Ipomoea sp.1*	*Indigofera linifolia*		*Mukia maderaspatana*
*Paspalidium sp.1**	*Mnesithea formosa*	*Ischaemum australis (?)*		Other grass'
Rhynchosia minima	*Scaevola sp.1*	*Melhania sp.1*		*Panicum decompositum*
*Schizachyrium fragile**	*Schizachyrium fragile*	*Mnesithea formosa**		*Schizachyrium fragile*
*Sida sp.**	*Sida sp.*	*Neptunia sp.1*		*Setaria surgens*
*Sporobolus sp.**	*Sida sp.1*	*Paspalidum sp.1*		*Solanum (echinolatum ?)*
*Tephrosia sp.2**	*Sida sp.4**	*Phyllanthus maderaspatens*		*Themeda australis*
*Tribulopsis sp.1**	*Sorghum sp.1*	*Portulaca oleracea*		*Themeda sp.*
Trichodesma zeylanicum	*Spermacoce sp.*	*Sporobolus australasicus*		*Tribulopsis sp.1*
*Triodia pungens**	*Sporobolus australasicus**	*Themeda australis*		*Triodia pungens**
Zornia muriculata	*Triodia pungens**	*Tribulopsis sp.1*		*Tephrosia sp.2*
Zornia sp.1		*Trichodesma zeylanicum*		
		*Triodia pungens**		
		Triodia sp.		

Table 3.4 (Cont)

Fire Age Class				
< 1 year post-fire	**1 - 2 years post-fire**	**2 - 3 years post-fire**	**3 - 4 years post-fire**	**> 4 years post-fire**
Shrubs				
*Acacia chisholmii**	*Acacia chisholmii**	*Acacia chippendalei**	*Acacia chisholmii**	*Acacia chippendalei**
Acacia galioides var galioides	*Acacia galioides var galioides**	*Acacia chisholmii**	*Acacia galioides var galioides**	*Acacia chisholmii**
*Acacia hilliana**	*Acacia hilliana**	*Acacia galioides var galioides**	*Acacia hammondii**	*Acacia elachantha*
Acacia holosericea	*Acacia shirleyi*	*Acacia gonoclada**	*Acacia hemignosta*	*Acacia gonoclada**
*Cleome viscose**	*Carissa lanceolata**	*Acacia hammondii*	*Acacia hilliana**	*Acacia hammondii*
Corchorus sp.*	*Corchorus* sp.*	*Acacia hilliana**	*Acacia monticola*	*Acacia hilliana**
Lysiphyllum cunninghamii	*Grevillea dryandra**	*Carissa lanceolata*	*Acacia phlebocarpa**	*Acacia holosericea*
Petalostigma quadriloculare	*Heliotropium* sp.	*Cleome viscose**	*Acacia tenuissima*	*Acacia monticola**
	*Petalostigma quadriloculare**	*Corchorus* sp.	*Acacia thomsonii (?)*	*Acacia retivenea**
	Senna sp.*	*Flueggea virosa**	*Carissa lanceolata*	*Acacia* sp.*
		Grevillea dryandri	*Jacksonia vernicosa**	*Carissa lanceolata**
		Hakea arborescens	*Melaleuca viridiflora*	*Corchorus* sp.*
		Maytenus cunninghamii	*Petalostigma quadriloculare*	*Crotolaria novae-hollandiae*
		*Petalostigma quadriloculare**		*Flueggea virosa*
		Scaevola ovalifolia		*Indigofera pratensis*
				Indigofera sp.
				*Petalostigma quadriloculare**
Trees				
Atalaya hemiglauca	*Acacia shirleyi*	*Atalaya hemiglauca*	*Acacia shirleyi*	*Atalaya hemiglauca*
*Corymbia bella**	*Corymbia terminalis**	*Corymbia terminalis**	*Atalaya hemiglauca*	*Corymbia* sp.1
Corymbia sp.1	*Eucalyptus grandifolia*	*Eucalyptus (grandifolia ?)*	*Corymbia terminalis**	*Corymbia terminalis**
*Corymbia terminalis**	*Eucalyptus leucophloia**	*Eucalyptus leucophloia)**	*Eucalyptus leucophloia**	*Eucalyptus aspera*
*Eucalyptus leucophloia**	*Eucalyptus leucophylla**	*Eucalyptus leucophylla**	*Eucalyptus leucophylla**	*Eucalyptus dichromophloia*
*Eucalyptus leucophylla**	*Eucalyptus pruinosa**	*Eucalyptus pruinosa**	*Eucalyptus* sp.	*Eucalyptus leucophloia**
Eucalyptus pruinosa	*Hakea arborescens*	*Eucalyptus* sp.		*Eucalyptus leucophylla**
Hakea sp.	*Terminalia canescens**	*Eucalyptus terminalis**		*Eucalyptus pruinosa**
*Terminalia canescens**		*Hakea arborescens*		*Grevillea striata*
		Hakea sp.1		*Lophostemon grandiflorus*
		Lysiphyllum cunninghamii		*Terminalia aridicola*
		Terminalia aridicola		*Terminalia canescens**
		Terminalia canescens		*Terminalia* sp.
				Ventilago viminalis

Regarding change in species diversity of apparent flora with time since fire, analysis of variance showed that there was a significant difference between the mean values of Simpson's Diversity Index at different fire ages ($F = 4.44$, $p < 0.003$, $n = 75$) (Figure 3.8). Mean species diversity was highest in the first two years post fire. The analysis of variance result was due to the difference in mean species diversity between fire ages one and two, and five (Tukey HSD, $p < 0.03$). Species diversity comparisons at other fire ages were not significantly different from one another.

When structural components were looked at separately, the diversity of ground cover species was correlated with fire age (Spearman's $R = 0.362$, $p < 0.001$, $n = 75$), with species diversity declining over time (Figure 3.9). The diversity of shrub and tree species did not change significantly post fire.

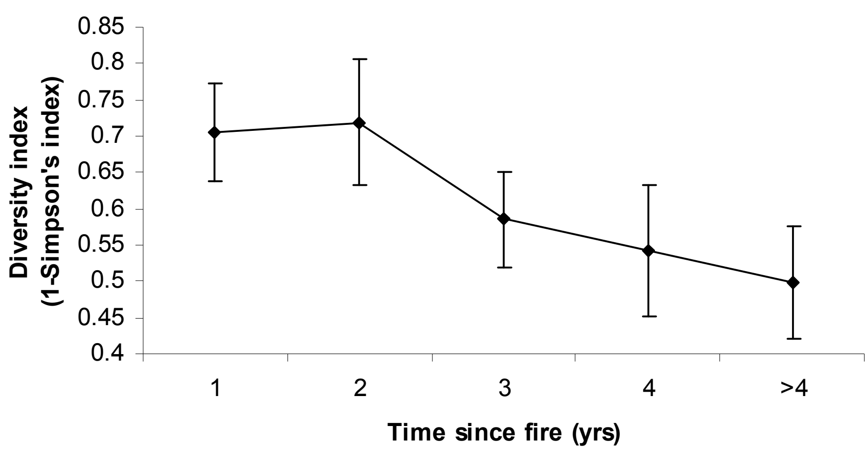

Figure 3.8 Species diversity declined over time (F = 4.44, p < 0.003, n = 75), driven by the difference in diversity in the first two years post fire (Tukey HSD, p < 0.03). (Bars are 95% confidence intervals).

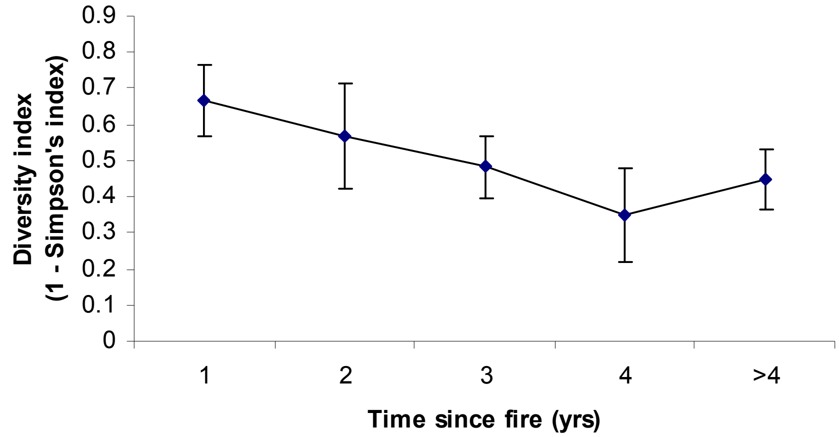

Figure 3.9 The diversity of ground cover species declined over time (Spearman's R = 0.361, p < 0.001, n = 75). (Bars are 95% confidence intervals).

The height diversity indices were calculated to provide a measure of vertical heterogeneity and to assist with determining whether the structure of the vegetation community changed significantly post fire. However, height diversity scores did not change significantly with time, either at the site level or when species were divided into ground, shrub and tree components. The absence of detectable change in the ground layer reflected the classification options available; mean ground cover for all sites was 0-50 cm deep, which only spanned two classes. Shrub and tree data spanned a greater range of height classes, but height variations were similar between recently burnt and recovering sites, suggesting that differences in height diversity scores were driven by site differences rather than post-fire recovery time.

Rank-cover curves suggested that there was a pattern to the changes occurring at the ground level (Figure 3.10), but when data for all strata were pooled, overall site diversity dominated the response. At the ground level, species richness decreased, and species dominance increased, with time since fire.

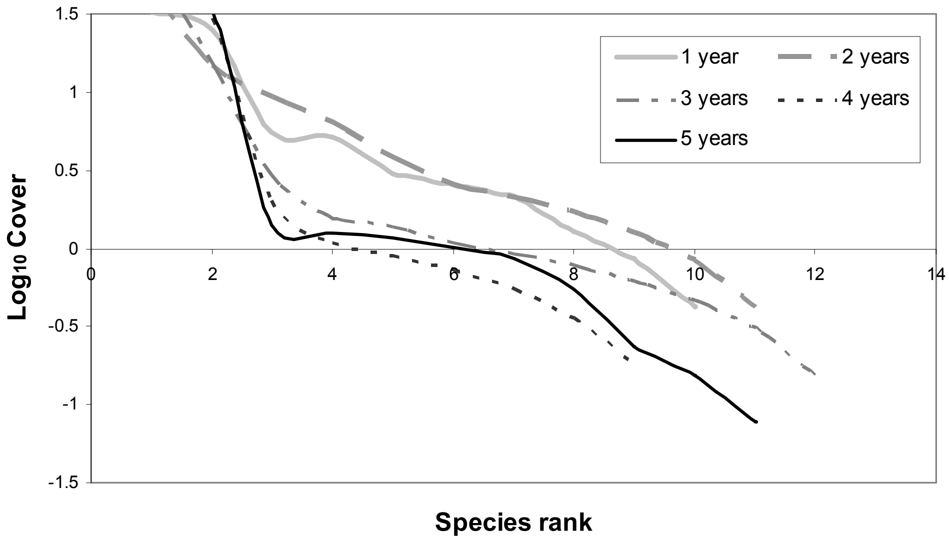

Figure 3.10 Rank cover curves (Gill 1999) for species in the ground layer showed that species richness decreased and species dominance increased in the years after fire.

Age classes and reproductive status

Ground layer

The demography of the ground cover changed with time since fire. The percentage occurrence of seedlings and juveniles decreased significantly with increasing time since fire (Spearman's R = -0.730, $p < 0.001$, n = 75), while the percentage occurrence of adult and old plants (Spearman's R = 0.730, $p < 0.001$, n = 75), and dead plants (Spearman's R = 0.278, $p < 0.02$, n = 75), increased (Figure 3.11). The proportion of ground cover that was reproductive showed a weaker response (Spearman's R = 0.230, $p < 0.05$, n = 75), however the strength of response was masked since annual and perennial plants were grouped together (see later).

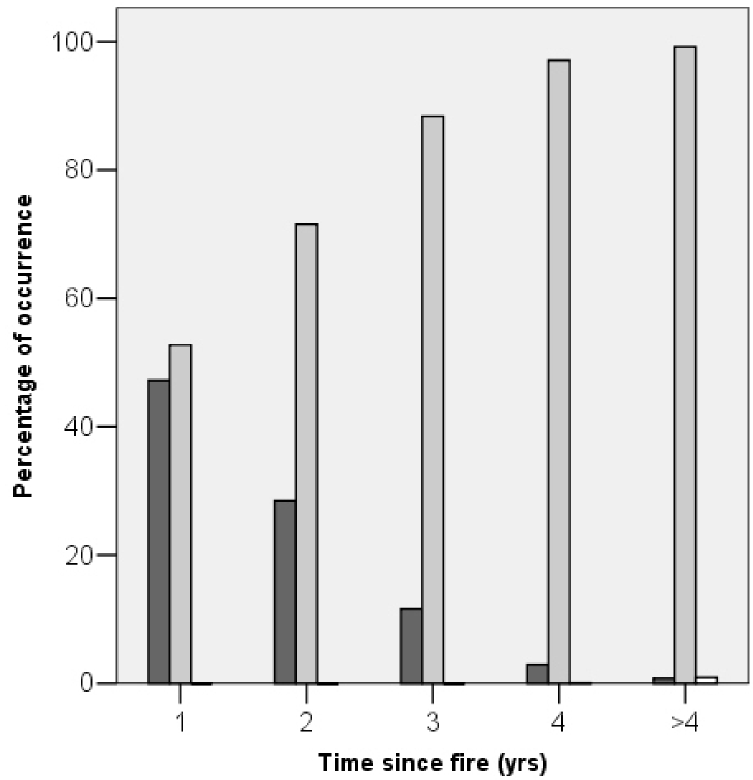

Figure 3.11 The percentage of ground cover plants that were seedlings or juveniles (dark bars) decreased with time since fire, with a concomitant increase in the percentage that were adult or old (pale grey bars). The percentage of dead plants (white bars) increased after fire-age four.

Shrub layer

For shrubs, the percentage occurrence of seedlings decreased with time since fire (Spearman's R = -0.277, $p < 0.05$, n = 58), but there was no correlation between the percentage occurrence of juveniles and fire age (Figure 3.12). The number of adult and old shrubs increased over time since fire (Spearman's R = 0.432, $p < 0.001$, n = 58), as did the proportion of shrubs reproducing (Spearman's R = 0.457, $p < 0.001$, n = 59). The occurrence of dead shrubs decreased over time (Spearman's R = -0.567, $p < 0.001$, n = 58).

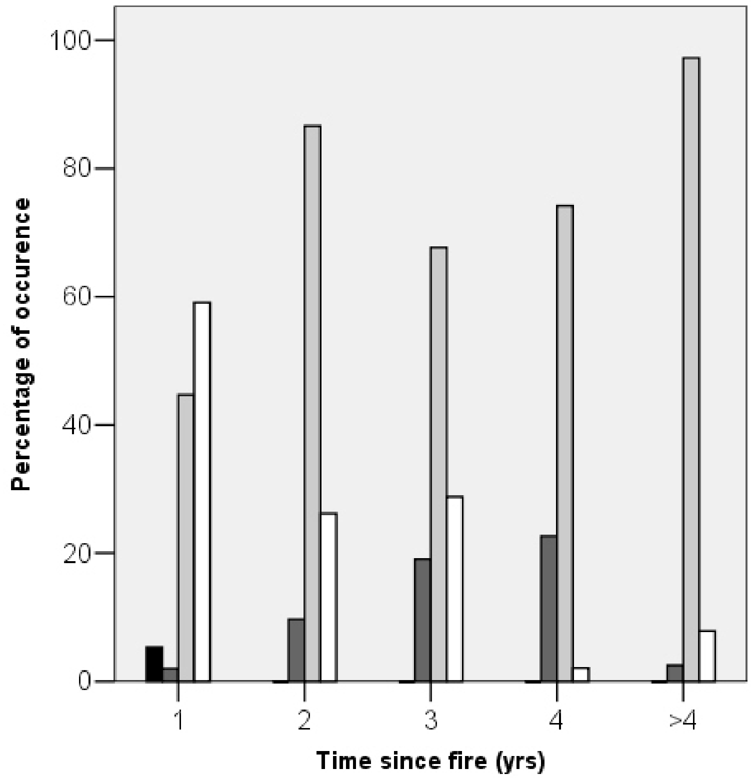

Figure 3.12 The proportion of shrubs at the seedling stage (black bars) or dead (white bars) decreased with time since fire, while the percentage of adult and old shrubs (pale grey) increased. The proportion of juvenile shrubs (dark grey) appeared to change non-linearly, but means were not significantly different at the $p < 0.05$ level.

Tree layer

The percentage occurrence of seedling, juvenile, adult or old trees was not correlated with fire age, however reproduction was, with more plants reproducing as time progressed (Spearman's $R = 0.417$, $p < 0.001$, $n = 74$). The percentage occurrence of trees with basal sprouts, trunk sprouts, or both, significantly declined over time (Spearman's $R = -0.425$, $p < 0.001$, $n = 74$; $R = -0.522$, $p < 0.001$, $n = 74$; and $R = -0.515$, $p < 0.001$, $n = 75$ respectively).

Functional groups

The complete species list and species' functional classification is provided in Appendix 2. Based on these data, the composition of the ground cover changed over time, with annual and ephemeral species more common in the early post-fire years than later (Figure 3.13), both in terms of percentage of species present and percentage cover. By five years post fire, perennial species accounted for almost

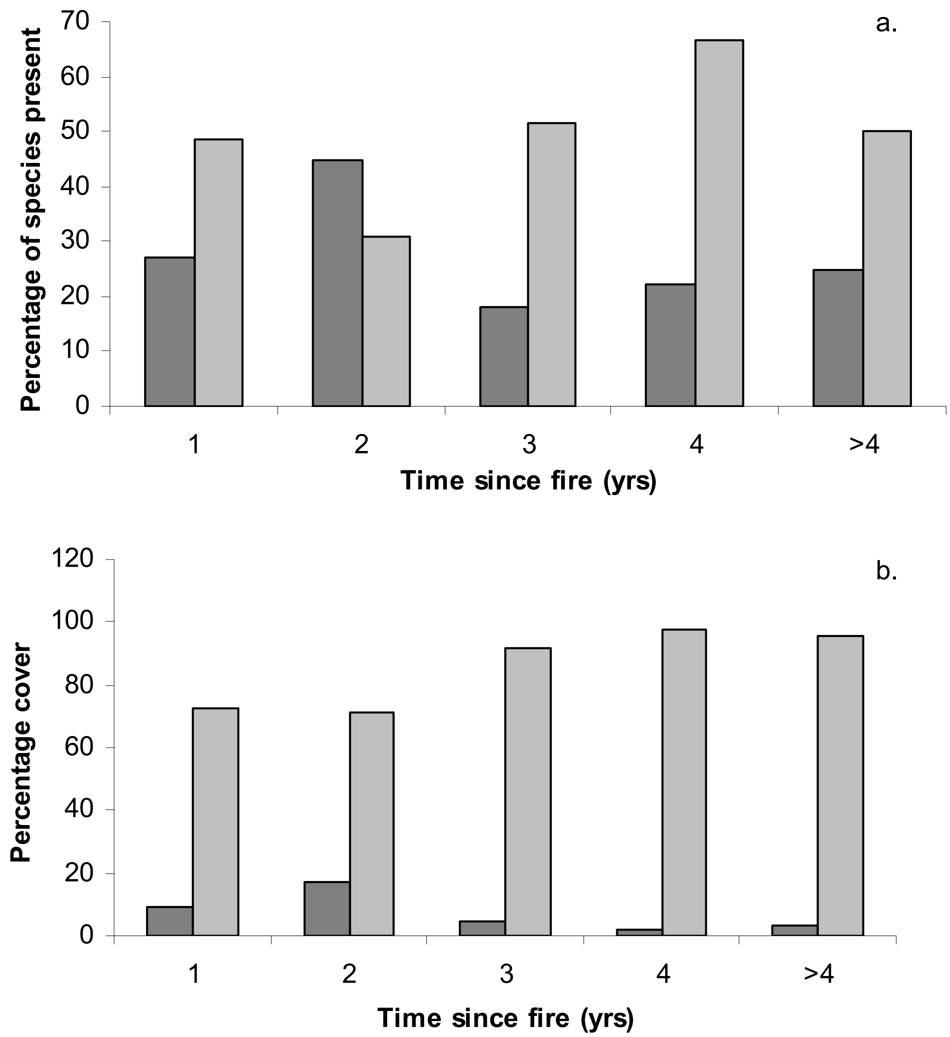

Figure 3.13 a) The percentage of ground layer species that were annual or ephemeral (dark bars) decreased over time. Light bars represent perennial species. b) The relationship was similar when percentage cover was considered.

100% of ground cover. Species that were obligate seeders showed a similar pattern (Figure 3.14), although there was much overlap in the species concerned, as most annual and ephemeral species were also obligate seeders. Obligate seeders were virtually gone from the ground layer by year three.

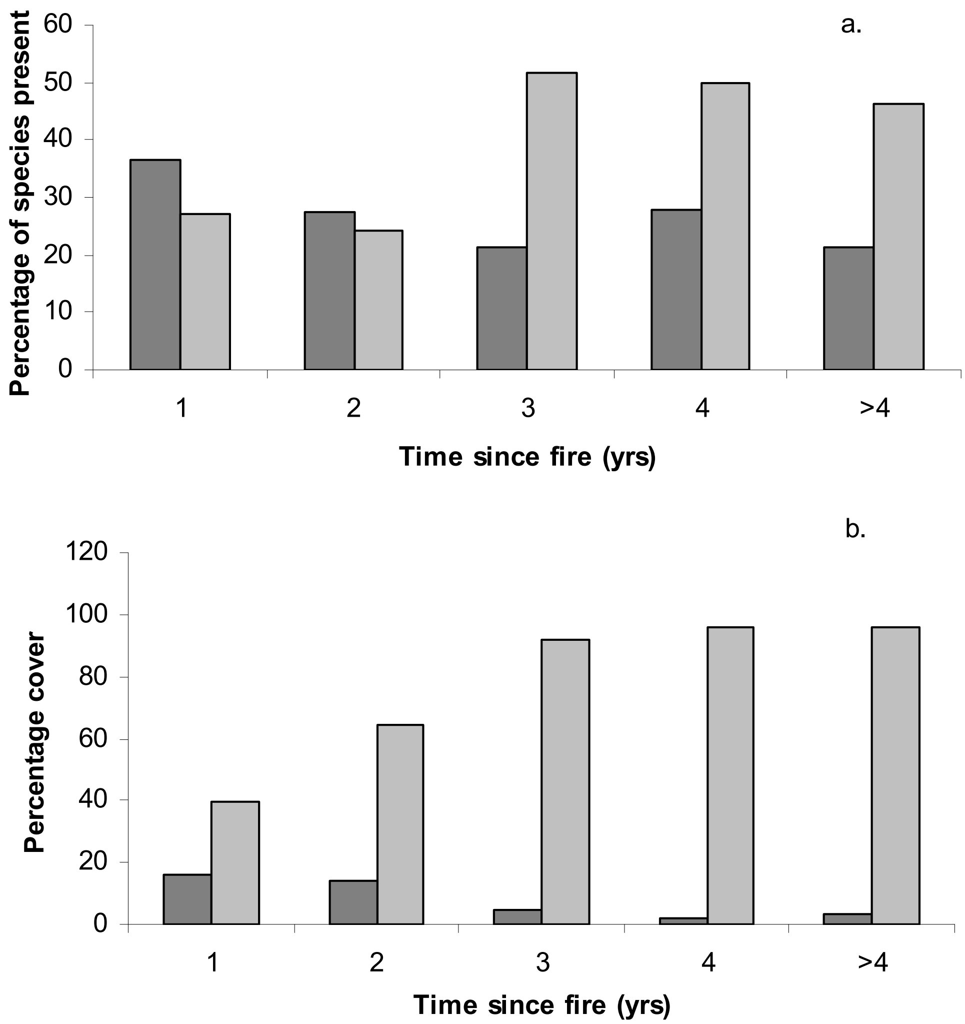

Figure 3.14 a) The percentage of ground layer species that were obligate seeders (dark bars) decreased over time, while representation of respouting species (light bars) increased. b) The relationship was similar when percentage cover was considered.

The percentage occurrence of perennial shrubs was relatively constant over the five years, but when classified into seeders and sprouters, the percentage of obligate seeders decreased over time, falling from 27.5% to 6.5% of the shrub species. All tree species were perennial sprouters, so no analysis was undertaken for this stratum.

Regenerative responses of the dominant species

Spinifex species, either hard or soft, dominated the ground cover at all sites. These perennial grasses showed similar recovery post fire. The percentage occurrence of seedlings and juveniles declined over time, while the percentage of grass that was adult or old increased (Figure 3.15). This was supported by data on reproduction, with seeding increasing significantly with time (Spearman's $R = 0.603$, $p < 0.001$, $n = 25$ for hard spinifex and $R = 0.571$, $p < 0.001$, $n = 49$ for soft spinifex). However, reproductive spinifex was present at all times since fire, as unburnt patches were present at each fire scar.

Acacia chisholmii was the dominant shrub at 20 of the sites. This species is an obligate seeder and adults are readily killed by fire. *Acacia chisholmii* regenerates after fire, with a flush of seedlings soon after rain. As the seedlings and juveniles mature, the percentage that is adult and old increases, while the percentage that is seedlings or juveniles decreases (Figure 3.16). Reproduction (and therefore adulthood) commences around three years post fire (Figure 3.17).

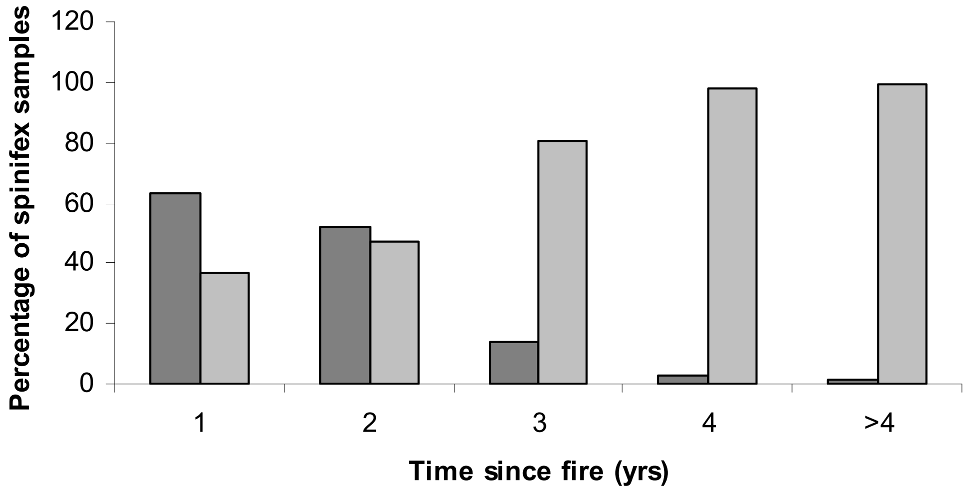

Figure 3.15 The percentage of spinifex that was at the seedling or juvenile phase (dark bars) decreased over time post fire, while the percentage that was adult or old (pale bars) increased. (Data for hard and soft species are combined).

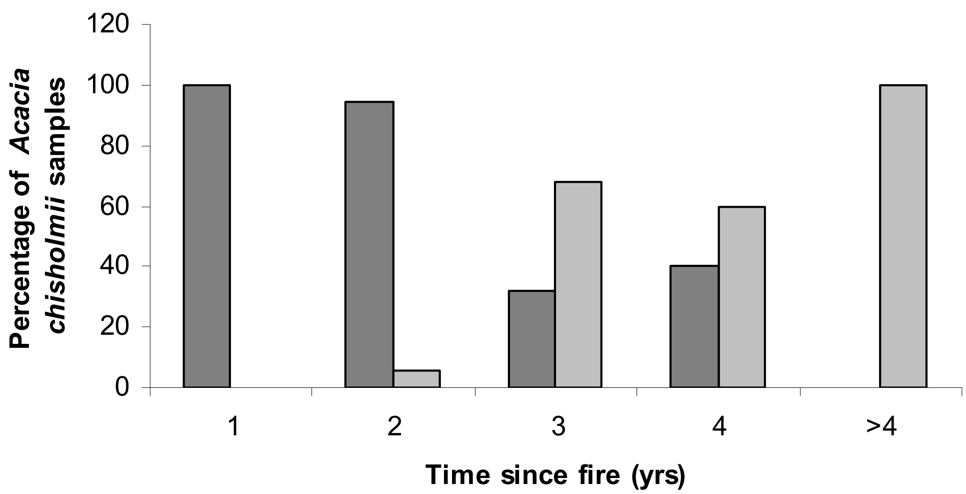

Figure 3.16 The percentage of *Acacia chisholmii* at seedling and juvenile stages (dark bars) decreased with time post fire, while the proportion of mature and old specimens (pale bars) increased.

No marked changes were seen in the demography of *E. leucophloia*. This species was dominant at 56 sites and sub-dominant at the others. Nearly all records were of adult or old trees; a small proportion of seedlings and juveniles were recorded at all fire ages. Reproduction increased as time progressed, however (Figure 3.18)

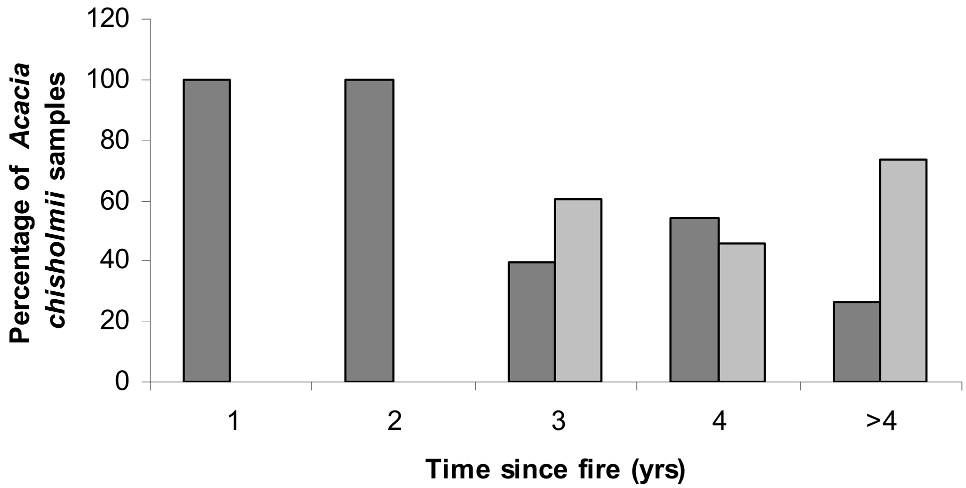

Figure 3.17 For *Acacia chisholmii*, reproduction commences around three years post fire. Pale bars represent the proportion of plants where buds, flowers or seed pods were present. Dark bars represent samples without reproductive material.

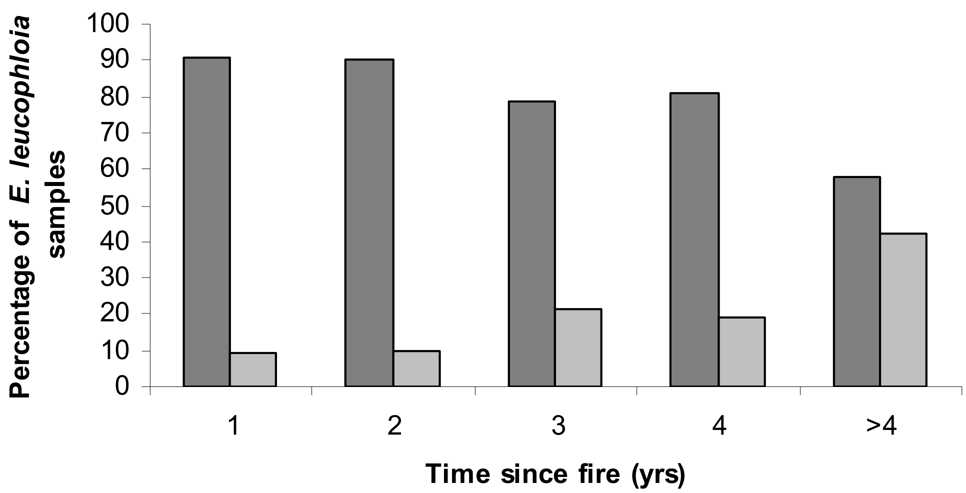

Figure 3.18 The percentage of *Eucalyptus leucophloia* that was reproductive (pale bars) increased with time since fire. Dark bars represent samples where reproductive material was not present.

3.3.2 Synthesis: Vegetation Dynamics in Spinifex/Snappy Gum Communities

The pathway of vegetation change in spinifex/snappy gum communities, and the recurrence of fire, was highly predictable (Figure 3.19). Fire completely removed most of the cover, but the vegetation was restored over time as species resprouted or were recruited from the soil seed bank.

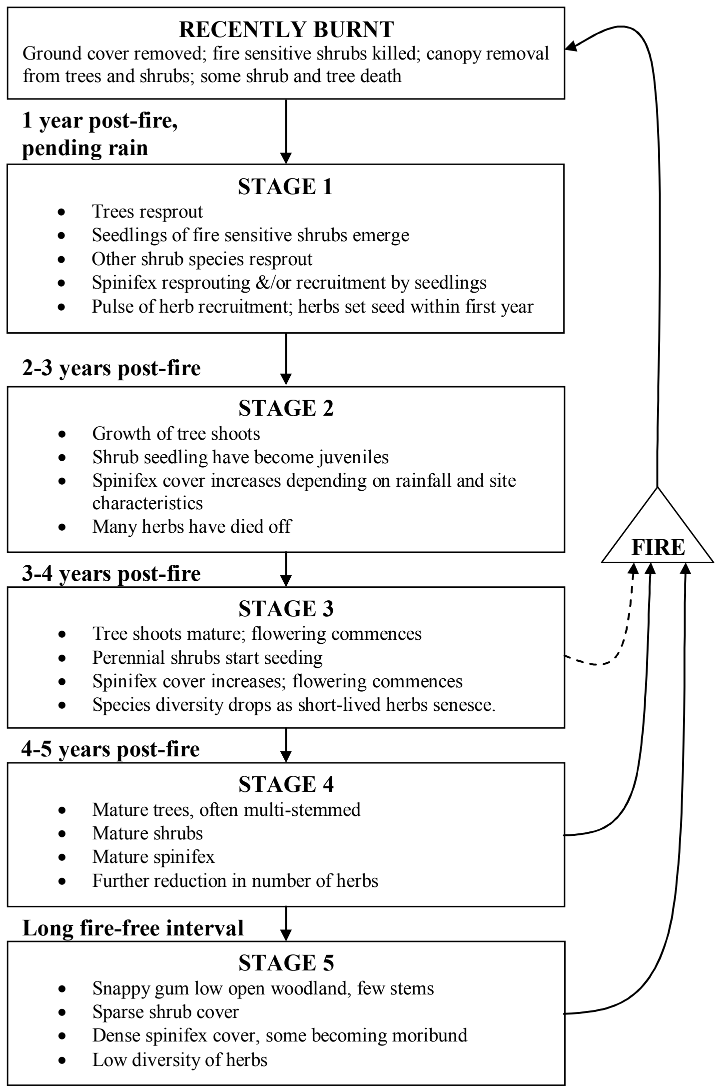

Figure 3.19 Fire completely removes most of the cover in spinifex/snappy gum communities, but the vegetation is restored over time as species resprout or are recruited from the soil seed bank. There can be sufficient fuel to sustain fire after three years, but fire intervals are usually longer.

3.3.3 Fuel Load Accumulation Over Time

Disc-drop calibration

On average, spinifex accounted for 79.1% of the total fuel load in each quadrat, coarse material made up 15.7% of fuel weight, and other grass and fine dicotyledons made up the remainder (5.2%). Data for each variable were transformed (Log_{10}) to improve normality, and for hard spinifex, hummock height was correlated to hummock weight (Pearson's R = 0.83, $p < 0.001$, n = 46) and hummock volume. The correlation between average hummock height and spinifex fuel load (Pearson's R = 0.75, $p < 0.001$, n = 46), was stronger than the correlation between percent cover and spinifex fuel load (Pearson's R = 0.72, $p < 0.001$, n = 46). When data from all sampling areas were pooled, disc-drop (fuel) height was positively correlated with total fuel load (Pearson's R = 0.853, $p < 0.001$, n = 106), spinifex fuel load (Pearson's R = 0.865, $p < 0.001$, n = 101) and coarse fuel load (Pearson's R = 0.227, $p < 0.05$, n = 88), but not the fuel load of dicotyledons. Also, the weight of coarse material increased as the fuel load of spinifex increased (Pearson's R = 0.333, $p < 0.002$, n = 88). These relationships held when data were examined separately for hard and soft spinifex species. The strength of these correlations confirmed that disc-drop height was suitable for estimating fuel load.

A simple linear regression using data from all sites was investigated for calibrating the disc-dropper. This was considered appropriate since vegetation data showed that hard and soft spinifex species frequently occurred together and the correlation coefficients were of similar magnitude. The model for Log_{10} drop height versus Log_{10} total fuel load was highly significant (F = 276.6, $p < 0.001$, df = 104), but for simplicity, the final calibration equation used untransformed data, which was slightly less accurate but more parsimonious (Figure 3.20):

Fuel load (t/ha) = (fuel depth - 6.87) / 2.33

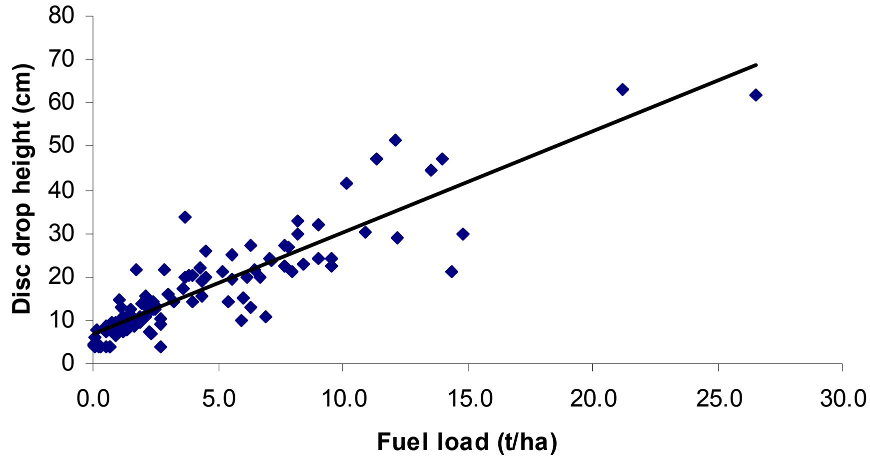

Figure 3.20 The relationship between disc-drop height and total fuel load was used to calibrate the disc-dropper ($R^2 = 0.80$, $p < 0.05$, n = 106). Points represent observed data; the line is the fitted regression.

Changes in fuel load over time

Spearman's Rank correlation coefficients were used to investigate changes in fuel load over time, as data for the 75 sites could not be easily transformed. Fuel load increased with time since fire across all sites (Figure 3.21) (R = 0.827, $p < 0.001$, n = 75) and when sites with hard and soft spinifex species were considered separately (R = 0.892, $p < 0.001$, n = 44; and R = 0.730, $p < 0.001$, n = 31, respectively).

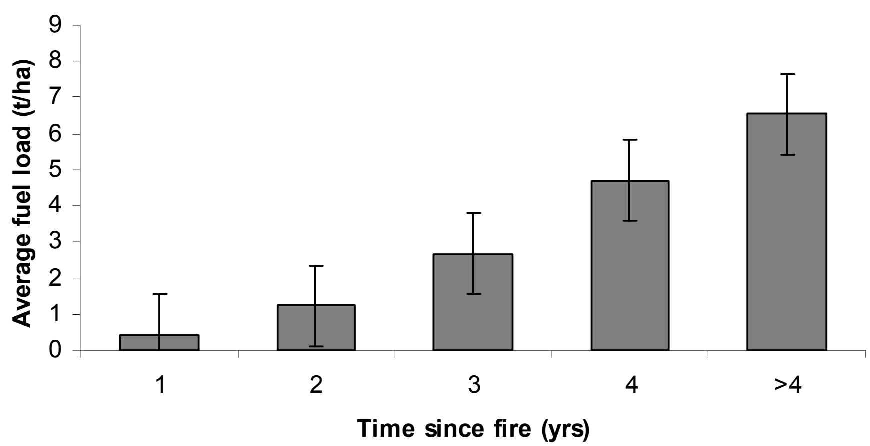

Figure 3.21 Fuel load increased with time since fire. Error bars show standard error (n = 75).

3.3.4 Variation in Fire Severity

Fire severity indicators

Broad scale comparisons based on data from 28 sites (56 transects) showed that canopy scorch, tree char height, shrub char height, % shrub charred and mean twig diameter were all positively correlated with each other (Table 3.5). The relationships held when data were analysed at the point scale (n = 1392). The values for R tended to be lower, but all were significant at $p < 0.001$.

Table 3.5. Tree and shrub variables measured as indicators of fire severity were positively correlated with one another (values are Spearman's Rank correlation coefficients, p < 0.001).

	% Tree canopy scorch	Shrub char height	Mean shrub twig diameter
Tree char height	0.57 (n = 56)	0.50 (n = 55)	-
% Shrub charred	0.75 (n = 52)	0.54 (n = 52)	0.69 (n = 52)

These indicators of fire severity were negatively correlated with the number of unburnt patches per metre. That is, the number of unburnt patches per metre was negatively correlated to tree char height (Spearman's Rank, R = -0.385, $p < 0.005$, n = 52), % canopy scorch (Spearman's Rank, R = -0.64, $p < 0.001$, n = 52) and % shrub charred (Spearman's Rank, R = -0.427, $p < 0.002$, n = 48).

Fire severity and patchiness indices

Fire severity was negatively correlated with fire patchiness (Spearman's Rank, R = -0.613, $p < 0.001$, n = 56). As transect fire severity scores increased, patchiness scores decreased. This was corroborated by point-based data where severity scores were negatively correlated with the number of unburnt patches (Spearman's Rank, R = -0.331, $p < 0.001$, n = 1392).

Variation in fire severity between and within fire scars

Analysis of variance showed that means of Log_{10} tree char height were significantly different between the 19 fire scars ($F = 31.4$, $p < 0.001$, $df = 18$, $n = 1391$). However, not all means were significantly different from one another. Fire scars could be grouped as homogeneous subsets (Table 3.6) and plotted along a continuum from low severity to high severity.

Table 3.6 Average tree char heights were significantly different from on another at the fire scar level ($F = 31.4$, $p < 0.001$, $df = 18$, $n = 1391$), although some fire scars could not be differentiated (Tukey's HSD test). Mean char heights grouped into homogeneous subsets allowed fire scars to be plotted along a continuum of fire severity.

Tukey HSD[a, b] — Log_{10} tree char height

Fire Scar No.	n	Subset for Alpha = 0.05: 1	2	3	4	5	6	7	8
5	28	1.9460							
13	37	1.9704							
4	44	2.0640	2.0640						
17	45	2.0906	2.0906	2.0906					
18	49	2.0960	2.0960	2.0960					
19	46	2.1161	2.1161	2.1161					
10	37	2.1582	2.1582	2.1582	2.1582				
14	48		2.2279	2.2279	2.2279	2.2279			
8	48		2.2559	2.2559	2.2559	2.2559	2.2559		
15	50		2.2596	2.2596	2.2596	2.2596	2.2596		
16	48		2.2780	2.2780	2.2780	2.2780	2.2780	2.2780	
12	47			2.3002	2.3002	2.3002	2.3002	2.3002	
11	48				2.3572	2.3572	2.3572	2.3572	
9	41				2.3590	2.3590	2.3590	2.3590	
7	47					2.3861	2.3861	2.3861	2.3861
2	47					2.4206	2.4206	2.4206	2.4206
3	44						2.4710	2.4710	2.4710
6	45							2.4873	2.4873
1	475								2.5851
Sig.		0.094	0.087	0.106	0.154	0.211	0.083	0.107	0.165

Means for groups in homogenous subsets are displayed
[a] Uses Harmonic Mean Sample Size = 45.731
[b] The group sizes are unequal. The harmonic mean of the group sizes is used.

The means of Log_{10} tree char height were also significantly different between the 28 sites ($F = 26.6$, $p < 0.001$, $df = 27$, $n = 1391$), but most fire scars had only one site. The ten sites within a single fire scar were significantly different from one another as well ($F = 35.08$, $p < 0.001$, $df = 9$, $n = 465$), showing that char height varied as much within a fire scar as between fire scars. This variation extended to a localised level. T-tests showed that 39.3% of the 'paired' transects had significantly different mean char height between the two pairs ($p < 0.05$).

There were weak but significant differences in Log_{10} char height between tree species ($F = 7.11$, $p < 0.001$, $df = 4$, $n = 1224$), and significant differences in Log_{10} char height between fire dates (month of burning) ($F = 24.934$, $p < 0.001$, $df = 5$, $n = 1391$). However, the fire timing data was not well replicated. January, February and March fires were only represented by one site each.

Analysis of variance results for Log_{10} twig diameter complemented the findings for char height. Log_{10} twig diameter varied between fire scars ($F = 10.822$, $p < 0.001$, $df = 18$), sites ($F = 11.63$, $p < 0.001$, $df = 27$, $n = 709$), month of burning ($F = 14.14$, $p < 0.001$, $df = 5$, $n = 709$), and shrub species ($F = 8.03$, $p < 0.001$, $df = 5$). Mean twig diameters were also significantly different between sites within a single fire scar ($F = 7.49$, $p < 0.001$, $df = 9$, $n = 153$). That is, twig diameter varied as much within a fire scar as between fire scars.

Kruskal-Wallis tests to compare severity scores between fire scars, between sites, and between sites within a single fire scar gave similar results to those obtained for individual char height and twig diameter measures. There were significant differences in severity scores between fire scars ($\chi^2 = 604.17$, $df = 18$, $p < 0.001$), between sites ($\chi^2 = 673.37$, $df = 27$, $p < 0.001$), and between sites within a fire scar ($\chi^2 = 126.07$, $df = 9$, $p < 0.001$). Mann-Whitney U tests showed that paired transects had significantly different severity scores at almost half the sites (46.4%) at the $p < 0.05$ level (Table 3.7). Likewise, the mean number of unburnt patches varied significantly between fire scars (χ^2 = 226.06, d f = 18, p < 0.001), between sites (χ^2 = 232.96, df = 27, p < 0.001), and between sites within a single fire scar (χ^2 = 24.32, df = 9, p < 0.005).

Table 3.7 Transect pairs had significantly different fire severity scores at almost half the sites (p < 0.05), indicating high within-site variability.

Transect pair	Mann-Whitney U	Probability	n
1, 2	69.5	**<0.001**	47
3, 4	67	**<0.001**	50
5, 6	230.5	0.098	50
7, 8	163	**0.003**	50
9, 10	175	**0.007**	50
11, 12	186	**0.013**	50
13, 14	303.5	0.86	50
15, 16	259.5	0.237	50
17, 18	207	**0.021**	50
19, 20	273	0.36	50
21, 22	283.5	0.546	50
23, 24	305.5	0.869	50
25, 26	94	**<0.001**	50
27, 28	56.5	**<0.001**	45
29, 30	257.5	0.273	50
31, 32	89	**<0.001**	50
33, 34	162	**0.002**	50
35, 36	295	0.68	50
37, 38	306.5	0.876	50
39, 40	310.5	0.968	50
41, 42	235.5	0.123	50
43, 44	238.5	0.142	50
45, 46	255.5	0.248	50
47, 48	4	**<0.001**	50
49, 50	58	**<0.001**	50
51, 52	149	**0.001**	50
53, 54	307	0.914	50
55, 56	253	0.24	50

3.4 Discussion

3.4.1 Vegetation Recovery After Fire

Post-fire vegetation changes in the spinifex/snappy gum communities of the Mount Isa Inlier were similar to those described for arid spinifex communities (see Allan & Southgate 2002; Gill 1999). Fire almost completely removes grass cover and tree canopies, and kills many shrubs. Most of the ground-cover (predominantly spinifex) and trees (predominantly eucalypts) resprout, and these sprouts rapidly grow over time, increasing cover. Many of the shrubs are obligate seeders and present within the soil seed bank. The seeds germinate to produce a mass of seedlings after fire. As a result, shrub cover is relatively stable after the first post-fire rains, with shrub cover composed of large number of small individuals initially and changing to a small number of larger individuals over time.

Species diversity changed over time. This supports the finding of Williams *et al.* (2002) where species diversity in the area's eucalypt woodlands increased in the year following fire, then decreased over time. In this study, species diversity peaked in the second year post fire, driven by an increased number of annual and ephemeral species in the ground layer. The above-ground diversity of ground cover species declined after the first year, whereas the diversity of shrub and tree species did not change significantly post fire. Trees and shrubs are perennial and remain in the community, even if reduced in size. Rank-cover curves confirmed the decrease in ground-level species richness, and increase in species dominance, with time since fire. It appeared that the numerical dominance of a few species prevailed over the considerable number of other species.

Height diversity scores were not significantly different between recently burnt and recovering sites, suggesting that score differences were driven by site characteristics rather than post-fire recovery time. The height diversity index did not accurately detect the visible variation in ground cover either, since most plants fell in the lowest two height classes. Greater diversity might have detected a finer scale, for example by assigning measures to 10 cm increments. Finer scale heterogeneity is likely detectable by ground dwelling fauna (York 1999) and may influence post-fire survival in some species.

Understanding changes in species' demography with time since fire helps with interpreting landscape dynamics, but all maturation stages were present in all strata, at all times since fire. Reproductive spinifex, for example, was always present because unburnt patches remained within each fire scar. Likewise, the percentage occurrence of juvenile shrubs did not change significantly over time. This suggested that germination continued in the years after fire, possibly mediated by rainfall. Regeneration can occur if the ground is bared by other means (e.g. senescence or death of other individuals), leaving a vacancy available for emerging seedlings (Bond & van Wilgen 1996). Not all regeneration responses are the direct result of heat/smoke/flame interactions with particular species. This variability is important because the persistence of plants in a landscape, or the resources available to fauna, might be underestimated by concentrating on fire scar age alone.

Investigating a variety of sites burnt at different times proved to be quick and convenient for assessing the vegetation community's response to fire. General fire-response characteristics could be determined at the landscape scale, although there were some limitations. The broad scale approach made it difficult to find study sites with uniform characteristics. Previous fire histories were not known (Gill 1999), nor were the conditions present at the time of burning (Whelan 1995). These shortfalls were considered acceptable, based on the premise that dominant patterns would be detected by visiting a large number of sites, and supplementary information would be provided by other aspects of the research. In the context of this study, the prime disadvantage of methods such as before-and-after studies, comparing adjacent burnt and unburnt plots, and observing *in situ* changes over time (Gill 1999) was the time required for sequential sampling.

Site stratification using a GIS was efficient for identifying potential study areas prior to field work, given the size of the bioregion and the scattered distribution of fires. However, it was constrained by coarse-scale vegetation mapping and out-dated road and bore data, even though this was the best available. It was difficult to find sites that met all five selection criteria. The 'haphazard' location of some sites was inevitable since predetermined sites could not be inspected prior to sampling and were sometimes inaccessible.

The wheel-point apparatus enabled rapid assessment of the dominant vegetation. The method is recommended for areas with sparse shrub and tree cover, particularly where data on the composition, cover and vertical structure of the vegetation are required (Friedel & Shaw 1987a, b; Griffin 1989). Indices of species diversity and spatial heterogeneity can be derived from the data, so relative comparisons can be made between sites. The main difficulty was in deciding what constituted a strike, although error was minimised by using a single observer (Griffin 1989). It was also difficult to categorise plants as annuals/perennials and seeders/sprouters without prior knowledge of the local vegetation. For example, some small woody herbs were short-lived perennials, often referred to as sub-shrubs (Vigilante & Bowman 2004). Their life span was longer than one year, but they were potentially similar to annuals in terms of fire response. There was uncertainty whether these low, compact sub-shrubs should be assigned to shrubs or ground cover, especially in the early post-fire stages. For other plants it was not always clear whether they were obligate seeders, resprouters or both, as insufficient time was spent searching for additional examples in the surrounding area. The intention was to supplement field data with information from the literature, but limited fire-response information was available. Plants that could not be identified to species level were precluded, so many of the responses recorded on the species list were ‘unknown’. More complete information is required before existing models, such as the vital attribute model of Noble & Slatyer (1978), can be confidently applied.

3.4.2 Fuel Load Accumulation Over Time

Fuel weight increased with time since fire, but did not follow the expected growth curve. The general pattern is for fuel to increase initially and then reach a steady state as growth rates match decomposition rates (Olson 1963; Walker 1981). The time until the mathematically-defined steady state is reached depends on growing conditions. The semi-arid climate restricts growth, and dead fuel has a relatively long turnover time (Gill *et al.* 1993), so equilibrium is expected to take longer than the 4 years reported for wetter eucalypt woodlands (Cook 2003; Walker 1981). The predicted asymptote may have been detected if fire history records covered a longer time period. The oldest fire age considered here was > 4 years, with data pooled for sites burnt five years ago or longer.

Fuel loads ranged from 0 t/ha to 26.5 t/ha (the latter at one site used for disc-drop calibration). The amount of fine fuel available was comparable to that reported for arid spinifex areas and mesic savanna woodlands. Cook (2003) reported that fuel load reached about 7.6 t/ha in eucalypt woodlands at Kapalga, Northern Territory; Craig (1993) estimated 6.4 t/ha in long unburnt sites of *Triodia pungens* in Western Australia; over 13 t/ha has been reported at sites in desert regions of Western Australia (Burrows *et al.* 1991 cited in Allan & Southgate 2002); and up to 40 t/ha has been reported for spinifex in Kakadu National Park, Northern Territory (Russell-Smith *et al.* 1998). Allan and Southgate (2002) speculated that the steady state in spinifex communities may be 8-10 t/ha, similar to the values obtained here at the longer unburnt sites.

There was substantial variation in fine fuel load at each fire age. The range was from none at recently burnt sites to 11.6 t/ha at longer unburnt sites. However, the maximum fuel load at one recently burnt site (up to 1 year post fire) was 3.8 t/ha, greater than the minimum fuel load (3.5 t/ha) at a site unburnt for > 4 years. This range in values at each fire age shows that past fire is not the only driver of fuel load. Some fuel can remain in unburnt patches within a fire scar and fuel accumulation is affected by site fertility, water availability, post-fire weather conditions and grazing impacts. Grazing has greater impact in the early post-fire years when young spinifex is palatable and the flush of other herbs is available (Craig 1993). These variables were not controlled for in this study. It is clear, however, that fuel increases over time, so more intense fires are expected at long unburnt sites.

Most methods for estimating fuel load, including the disc pasture meter (Bransby *et al.* 1977), were developed for continuous grass swards. It was uncertain whether the ‘disc-dropper’ would work for discontinuous hummocks of spinifex. Craig (1993) used a pasture depth metre for estimating fuel load of *Triodia pungens* in Western Australia by calibrating fuel load against the compressed height of individual hummocks. This was to overcome discrepancies where leaf blades extended beyond the main hummock envelope. However, there were issues with discriminating individual hummocks when they became large and entangled and it was not successful when hummocks were very small, 1 to 2 years post fire. In this study, disc-drop measures were based on a random walk and included bare ground, immature hummocks, partial hummocks and full hummocks. This allowed for ‘site averaging’ and overcame difficulties in selecting which hummocks to measure.

Fuel load was strongly correlated to disc-drop height and the method was quick and easy to apply. Issues such as deciding the exact measure when the ground was uneven or rocky, or when the disc rested at a sharp angle on a hummock edge, were addressed by consistently reading measurements where the disc was in contact with the rod. Correlation strength for total fuel load (coarse and fine material) across all sites was slightly less than that obtained by Craig (1993) for individual hummocks of *Triodia pungens*. This was likely due to the species mix and variations to the method, but issues associated with calibration need to be considered. Fuel harvesting, weighing, quadrat positioning and size can also introduce inaccuracies (Catchpole & Wheeler 1992). The relationship between fuel load and disc-drop height was considered sufficiently strong for broad scale comparisons between sites. The main disadvantage of the method is the need to recalibrate if discs of different weight and area are used, or if the level of fuel curing affects resting height (Craig 1993), which was not investigated here.

3.4.3 Variation in Fire Severity

Fire severity varied as much within fire scars as between fire scars. Mean tree char height, shrub twig diameter and number of unburnt patches per metre varied significantly between fire scars, sites, sites within a single fire scar and between transects (at almost half the sites). Leaf scorch, tree char height and shrub twig diameters were collinear, which provided some assurance that they were suitable indicators. It also meant that fire severity scores, which were derived from these measures to provide an index for comparison, also varied.

The high level of variability within a fire is well known (Atkins & Hobbs 1995; Lindenmayer & Burgman 2005; Tolhurst 1995; Whelan 1995), but not well documented, despite its importance for the survival of organisms and ecosystem functioning (Atkins & Hobbs 1995; Whelan 1995). Many factors contribute to intra-fire differences in fire severity. They include: topography, variation between species' combustibility, variable wind speeds, variable litter cover, fuel conditions (moisture content, packing, composition, continuity), temperature at time of burning and more (Catchploe 2002; Whelan 1995). In hummock grasslands, the great spatial variability in fuel conditions alone can help explain the variation detected (Gill *et al.* 1995). In one small area, cover can vary from 100% (for example in drainage lines) to near 0% (on rocky outcrops). Curing is variable, with green, cured and moribund grasses present at the same time. Hummocks vary in size, density and shape, and are separated by variable stretches of bare ground. Gill *et al.* (1995) considered these factors in terms of their effect on fire spread; through this they affect variation in fire intensity and help to explain site heterogeneity.

Fire severity measures varied according to the species present as well. Mean char height was significantly different between tree species, possibly due to their burning characteristics or flammability (e.g. volatile oils, bark structure). Few sites were pure stands, so different species mixes contribute to intra-site variability. Shrub data showed a similar trend, with some variability in twig diameter being attributable to shrub species.

Fire intensity is commonly linked to the season of burning. In Australia's tropical savannas, late dry season fires are more intense than fires lit early in the dry season or wet season (Williams *et al.* 2003b). My results show significant differences in tree char height with respect to month of burning, but the fire timing data were not well replicated. Site location was *ad hoc* and depended on finding accessible areas that had burnt within the previous six months. There were no controls over variables such as fuel conditions, weather or topography, so the influence of fire season could not be addressed. Fire scars could be plotted on a continuum of fire severity, however, even with the high degree of intra-site variability. This suggests that post-fire severity measures could be useful for quantitatively assessing the outcome of planned fires.

Variability in fire severity was detected at a finer scale than was visually apparent, as found by Tolhurst (1995). Few of the fire scars visited looked 'patchy'. Most appeared to have been intensely burnt and were not visually distinctive from a fire severity perspective. This finding could be useful for monitoring the outcome of planned fires, where the goal is to promote fire regime heterogeneity. An index could be used for comparing fire severity between different sites or at the same site over time. The index derived in this study needs testing and refinement, but conceptually, using the median of a number of scores overcomes difficulties in comparing sites where there are noticeable differences in vegetation. For example, if only one variable such as twig diameter had been measured, sites with or without a shrubby understorey could not be conveniently compared. Severity and patchiness indices were negatively correlated with each other, supporting the validity of the approach and potentially allowing fire 'patchiness' to be gauged. Research to explore variables that would most usefully contribute to an index, the most appropriate formula for combining them, the level of variability that can be detected, and how transferable the index might be between vegetation types, is recommended.

Measuring surrogates that relate to fire intensity, such as leaf char height, leaf scorch height, cover and shrub twig diameters (Whelan 1995; Williams *et al.* 2003b) are less resource intense than directly measuring fire intensity by calculating fuel loads, fire spread rates and heat of combustion (Williams *et al.* 2003a). They are convenient for ecological studies such as this, where post-fire site inspections preclude techniques such as measuring evaporative water loss (Tolhurst 1995) or colour changes to heat sensitive paint (Atkins & Hobbs 1995) that are possible for experimental fires. Protocols were required to ensure consistency, however. For example, some fire scars had aged six months by the time they were sampled, so plants were beginning to recover. Canopy scorch of trees and shrubs could still be discerned, but care was required to discriminate newly flushed leaves from old leaves. Estimates of char height were not always straightforward. Some specimens of *E. leucophloia* had shed their bark. This species also appeared to have two char heights, possibly due to the propensity of their oil-rich canopies to ignite without direct contact with flame (Ron Croft pers. comm., Bob McDonald pers. comm.).

These issues were discussed as they arose and data collection procedures were agreed with by both sampling teams.

3.4.4 Vegetation Dynamics in Spinifex/Snappy Gum Communities

The fire response of spinifex woodlands in the Mount Isa Inlier was similar to arid-zone spinifex communities, but recovery occurred over a shorter timeframe than reported for areas with intermittent rain (Allan & Southgate 2002). The pathway and rate of vegetation change are more predictable in this bioregion given the higher, more consistent rainfall.

Two aspects of the system make it resilient to fire. Fire cannot usually spread for at least three years from the previous fire (and then only in very hot, windy conditions) because there is insufficient fuel. By this time, most herbs have set seed and senesced, and shrubs, trees and spinifex have started to reproduce. Secondly, the inherent variability within the burnt area means that patches of vegetation at later recovery stages co-exist with vegetation at earlier stages. Mature plants in these patches provide reproductive material that helps to ensure that species remain in the system.

The longer term effects of the fire regime need to be considered for adaptive management, with consideration given to fire frequency, intensity and season of occurrence (Andersen 2003). In the Mount Isa Inlier, fire frequency is limited by the time it takes for fuel to accumulate. It is anticipated that a three to four year fire return time would retain the majority of plant species, but community structure would change. Tree basal area and cover would be reduced and trees would be predominantly multi-stemmed. This fire frequency would also diminish the cover of fire sensitive shrubs as the soil seed bank was gradually depleted (but not exhausted). If this fire frequency was sustained, which is unlikely given the complex landscape, tree seedlings would not be recruited to the canopy.

The overall fire intensity appeared less critical to plant population dynamics in the Mount Isa Inlier than in other areas. Fire severity varied between fire scars, but all fires were relatively intense, completely removing the canopies of many trees. It appeared that if spinifex communities were ignited, they burnt fiercely or did not burn at all, similar to heath communities (Peter Stanton pers. comm.) and chaparral fires in North America (Lindenmuth & Davis 1973 cited in McCaw 1995). Small fires of lesser intensity occurred, but limited fire spread precluded sampling using the wandering quarter method. In contrast, fires in eucalypt woodlands of the coastal tropics and mesic savannas range from low intensity fires that remove ground cover with little impact on trees and shrubs, to higher intensity fires with significant canopy scorch (Williams *et al.* 2003b). Fires of low intensity appear difficult to achieve in spinifex country, but the inherent variation within a fire scar will guarantee fire regime heterogeneity over time.

The interplay between the seasonality of fire and response of vegetation was not investigated here. Williams *et al.* (2003b) found that more intense, late dry-season fires reduced flowering in five out of six tree species, but composition of the ground layer was driven by variation in annual rainfall. It is anticipated that the almost complete removal of tree and shrub canopies in spinifex/snappy gum communities will delay flowering, regardless of the season of burning. In the ground layer, composition is determined by the soil seed bank and seedlings emerge after rain, independent of when the fire occurred. The timing of fire may indirectly affect ground layer composition, however, if there is insufficient follow-up rain to sustain seedling growth.

3.4.5 Implications for Management and Research

A better understanding of landscape dynamics assists with setting and monitoring fire management objectives. The information presented above suggests that spinifex/snappy gum communities are resilient to fire. Fire return intervals are limited by fuel load, which does not accumulate sufficiently until after four or more years when most species have reproduced or commenced reproduction. Nonetheless, frequent burning (every 4 - 5 years) is predicted to result in structural changes to the vegetation. This may be the objective for some managers (e.g. the removal of *Acacia chisholmii* for easier livestock management; Peter McNeven, pers. comm.), but not if landscape heterogeneity is sought. In this case, visual clues can be used to assess the landscape and guide the frequency of burning. For example, the stem sizes of eucalypt trees could be used as an indicator. Snappy gums with a single trunk, or a small number of stems of reasonable size, indicate a significant time since fire. Re-burning could be considered, being careful to ensure that some areas remained long unburned. Broad areas of snappy gum that take the mallee form indicate more recent, or too frequent, burning and should be left for longer. Other indicators include the proportion of senescent spinifex, or age structure of fire sensitive shrubs. If habitat heterogeneity is the objective, a range of fire-recovery stages should be present in the landscape.

A second concern of managers is 'how much area to burn?'. This is a key question in the Mount Isa Inlier. Fires of low intensity are difficult to deliver; but fine-scale heterogeneity is somewhat assured due to intrinsic variability within fire scars. This gives some justification for specifying the desired fire extent as a management objective, rather than a degree of internal patchiness. But this needs to be tempered with speculation that internal patchiness will affect the long term persistence of species (Atkins & Hobbs 1995; Lindenmayer & Burgman 2005; Whelan 1995). An ability to quantify landscape patchiness would assist in testing this concept. Landscape patchiness and optimal fire size are addressed in subsequent chapters.

Understanding vegetation dynamics may contribute information to other conservation initiatives. For example, legislation to protect rare or threatened plants is based on species lists compiled from herbarium records. Rarity and threat are assigned through expert opinion, with lists reviewed regularly to reflect changes to the information base. Collection effort can skew the lists (Dickman *et al.* 2004), which may be of particular concern in this semi-arid area where fire scars might be considered depauperate of plants. The flush of herbs after fire can elucidate species thought to be rare (Keith McDonald, pers. comm.). Research is required to identify genuinely rare species that require management attention; targeting fire scars may help.

Finally, fire management objectives are not based on vegetation alone. My findings suggest that a 5-year fire return interval will provide for plant persistence, but the impact of fire on resident fauna has not been considered. It takes time for recovering vegetation to be suitable for some animals to recolonise. Vegetation at year 5, for example, may just be becoming suitable for fauna species such as Grasswrens and Spinifexbirds to re-establish (see chapter 5). Frequent burning would remove this option.

3.5 Conclusion

I found that vegetation recovery follows a general and predictable pattern that can assist managers to anticipate the response of different fire regimes. There is a flush of ephemeral species after fire, but for much of the time these species are hidden in the soil seed bank. Most perennial species demonstrate a high level of resistance to fire by sprouting. Fire-sensitive perennial species are able to reproduce before subsequent fires can be sustained, given the sufficiently slow rate of fuel load accumulation. There is substantial fire-induced heterogeneity at landscape and local scales, which is potentially significant for conserving species. The variation in severity within fire scars provides an argument for concentrating on fire size as a management goal.

Suggesting an optimal fire size requires further research. First, the issue of implementation arises; understanding the distribution of fuel in the landscape is required. Second, determining whether variability immediately after fire is similar to overall landscape variability would provide insight to the degree that fire influences the system. Finally, the link between internal fire patchiness and the persistence of fauna needs to be explored. Nonetheless, the above work provides baseline information on the response of vegetation to fire in the Mount Isa Inlier, which is critical for fire planning and monitoring.

4. UNDERSTANDING PATTERNS: MAPPING FUEL LOAD AND FIRE SEVERITY

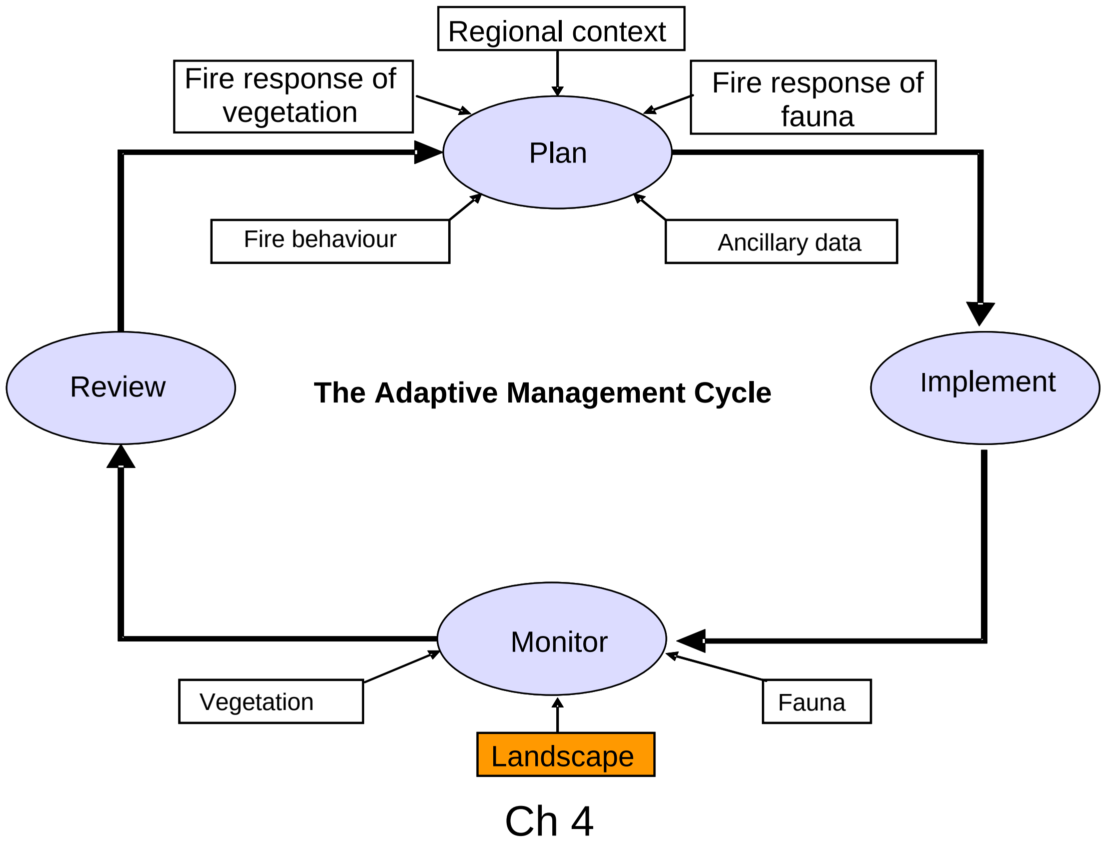

Figure 4.1 Fire planning objectives related to landscape patchiness can be evaluated by monitoring broad scale fire patterns. Such patterns also aid understanding of pattern/process interactions.

4.1 Introduction

Ecological fire management requires information from species to landscape scales. Ground-based studies show that the response of vegetation to fire depends on fire intensity, which varies at a localized scale due to changes in fuel load, fuel moisture, fuel type and arrangement, terrain, wind and relative humidity (Catchpole 2002; Whelan 1995). In addition, factors controlling landscape pattern need to be understood to predict vegetation development and understand ecosystem dynamics (Turner *et al.* 2001). Links are required between information on the fire response of individual plants, to the persistence of plant populations and meta-populations. But extrapolating from the local level at which plant response data are collected, to the landscape level where fire operates as an ecological process, is a recurrent problem. A mismatch exists between the fine scale heterogeneity that occurs on the ground and the broad scale at which fire mapping is typically undertaken. 'Average' temperatures are reported in fire intensity studies (Whelan 1995) and, by necessity, maps of fuel load are generalised so overlook within-stand variation (Keane *et al.* 2001). This limits ecological interpretation. Since spatial complexity within a fire footprint can be significant to plant population persistence (Ooi *et al.* 2006), documenting or quantifying landscape variability will contribute to the successful management of species. Mapping fuel load provides a means to do this and assists with understanding the drivers of landscape pattern, thus contributing directly to fire planning (Figure 4.1). It also plays a role in implementation, especially if an optimal fire size is sought, as hazardous zones or areas of likely fire spread can be identified.

Remotely sensed satellite data have been applied to fire studies and have helped to identify patterns and address issues of scale. Information on fire extent and fire timing is routinely obtained from a number of different satellites (Justice *et al*. 2003), which allows fire extent, frequency and seasonality to be documented. In Australia, such information has been collated at national and regional scales using data from NOAA-AVHRR and Landsat satellites (1-km and 15 to 30-m pixel resolution respectively) (Craig *et al.* 2002; Russell-Smith 2002; Smith *et al*. 2000). Maps prepared from these data are used for fire planning and monitoring (Justice *et al*. 2003), and measuring and monitoring broad-scale vegetation change (Danaher *et al.* 1998). However, a mismatch still exists between the fine scale heterogeneity that occurs on the ground and the scale of imagery currently used for mapping. Fires are internally more variable than is indicated by Landsat-derived fire scars (Price *et al.* 2003).

Mapping applications using new generation, high-resolution satellites such as IKONOS (1-m – 4-m pixel resolution) provide an opportunity to bridge the ground-aerial gap. Use of this finer scale imagery is at the developmental stage regarding ecological problems, including fire studies. It should, however, provide additional insights for determining the impact of fire on vegetation and documenting fire-induced heterogeneity.

The benefit of satellite imagery over other remotely sensed data, such as aerial photography, is that quantitative data are potentially available for several months each year, and over a range of different wavelength bands of radiation. These data can be manipulated to enhance different aspects of the Earth's surface and can be statistically interrogated (Burrough & McDonnell 1998). Satellite imagery is particularly suitable for analyzing continuous data, such as along environmental gradients, and specific analytical methods have been developed for this purpose (Burrough & McDonnell 1998; Fortin & Dale 2005; Gustafson 1998).

Uses for satellite-derived data and associated methodologies have been developing steadily since the launch of the first remote sensing satellites in the early 1970s. Techniques continue to evolve as image resolution becomes finer and data is captured on a greater number of bands of the electromagnetic spectrum. Studies of vegetation commonly use data from the green, red and near infrared (NIR) bands, either individually or combined mathematically into indices, with digital map data correlated with field data. Most indices were developed from northern hemisphere studies, however, and there are concerns that the usual assumptions may not apply in the Australian rangelands (Graetz & Gentle 1982).

The aim of this chapter is to investigate the application of fine-scale satellite imagery (IKONOS) for mapping fuels and fire severity in northwest Queensland. The approach builds on the information presented in Chapter 3. Emphasis is on fuel load and fire severity because these complementary variables strongly influence plant responses to fire. Mapping these two components will assist in scaling up from site-based measurements to a broader scale. Also, satellite imagery provides an alternative tool for investigating landscape patterns. Additional insight on fire/vegetation dynamics in the Mount Isa area is anticipated.

The approach taken is exploratory. Field data on fuel load and fire severity were compared to reflectance values recorded by the IKONOS satellite, as well as a number of derived indices. Relationships between field measurements and reflectance values were investigated to identify trends with respect to fire age, ground cover type, tree type and landscape position. Strong associations between field and satellite-derived data allow models to be developed which can then be applied across entire images to produce thematic maps (Keane *et al*. 2001). These maps offer potential for predicting fuel load or fire severity at different locations. They also enable the links between fuel load and fire severity to be examined further and provide a foundation for studies related to spatial heterogeneity pre- and post fire. For example, the amount of fuel available or burnt, and the severity of the burn, can be investigated and the resultant variability quantified.

Mapping fuel load and fire severity using IKONOS imagery would be a significant advance. Fuel load mapping from satellite imagery is more common in the United States than Australia. The focus there is on national, regional and sub-regional levels with emphasis on risk assessment rather than ecological outcomes (Burgan *et al.* 1998; Keane *et al.* 2001; Schmidt *et al.* 2002). The scales used are appropriate for evaluating general vegetation condition for regional planning (Morgan *et al.* 2001), or developing models to simulate fire growth and intensity (Keane *et al.* 2001). However, they are too coarse for interpreting the impact of fire on plant species or recording spatial variation at a scale that may be relevant to fauna. Finer scale mapping would give accurate and spatially explicit fire histories for different areas, which would benefit researchers interested in characterising fire regimes. Mapping fire severity at this scale would be particularly notable. Fire intensity is the key aspect of fire regime not yet captured by remote sensing (Hammill & Bradstock 2006) and this factor has limited interpretation of vegetation response (Russell-Smith *et al.* 1998). Detecting fire intensity directly from satellites is still experimental (Justice *et al.* 2003) and is likely to be problematic for routine fire mapping (Hammill & Bradstock 2006). Post-hoc fire severity mapping is an alternative approach but analytical methods need to be established (Hammill & Bradstock 2006).

The chapter uses conventional image processing and analysis techniques. These have not been well tested for use with IKONOS imagery. Issues such as image pre-processing are still under debate (Law & Nichol 2004) and guidelines for integrating field survey information with remotely sensed data are still being developed (Reinke & Jones 2006). IKONOS imagery is yet to be applied in a wide range of environments, resulting in uncertainty whether common assumptions are valid for different applications and areas. Investigating the application of IKONOS imagery to fire management in the semi-arid Mount Isa Inlier will inform these debates. Due to the exploratory nature of this work, each method is presented and evaluated, with the discussion focussing on general issues related to fine scale satellite imagery and the implications of these findings for research and management.

4.2 Methods

4.2.1 The Study Area

The study was undertaken in two separate areas, roughly in the centre of the Mount Isa Inlier (Figure 4.2). Both areas were located on Calton Hills station which covers approximately 393,000 hectares of dissected terrain. The dominant vegetation is low open eucalypt woodland with a spinifex understorey. Eucalypt species include *Eucalyptus leucophloia*, *E. leucophylla*, *E. pruinosa*, and *Corymbia terminalis*. Spinifex species include *Triodia longiceps*, *T. molesta*, *T. bitextura* and *T. pungens*. Other annual and perennial grass genera, such as *Schizachyrium, Aristida, Heteropogon, Dichanthium* and *Chrysopogon,* occur along drainage lines and on river flats. A shrubby mid-storey is frequently present, dominated by species of *Acacia*, especially *A. chisholmii* and *A. hilliana*. Pure stands of *A. cambadgei* and *A. shirleyii* occur as pockets of woodland known as ‘Gidgee’ and ‘Lancewood’ respectively.

Pastoralism is the predominant land use. Cattle graze on native species, although the introduced pasture grass *Cenchrus cilliaris* (buffel grass) has been sown in some areas. Mining activity is also present, especially in the western part of the study area where there is an active copper mine. There is evidence of past mineral exploration throughout the region, including drill-rig pads, mullock heaps and wheel tracks. This disturbance is patchy and the majority of the area is covered with native vegetation.

The Mount Isa Inlier is characterised by rocky, broken country. In general, access is poor. The two areas considered here were chosen because the vegetation was representative of that occurring throughout the bioregion, a range of different fire ages was present and there was reasonably good access. Areas of different fire-age were considered important for fuel load mapping because time since fire is a key driver of fuel load. There were north-south access tracks on both sides of the larger, more southerly area, and a power line bisected it, enabling sampling from a number of points. No human settlements were present within the areas depicted by the imagery.

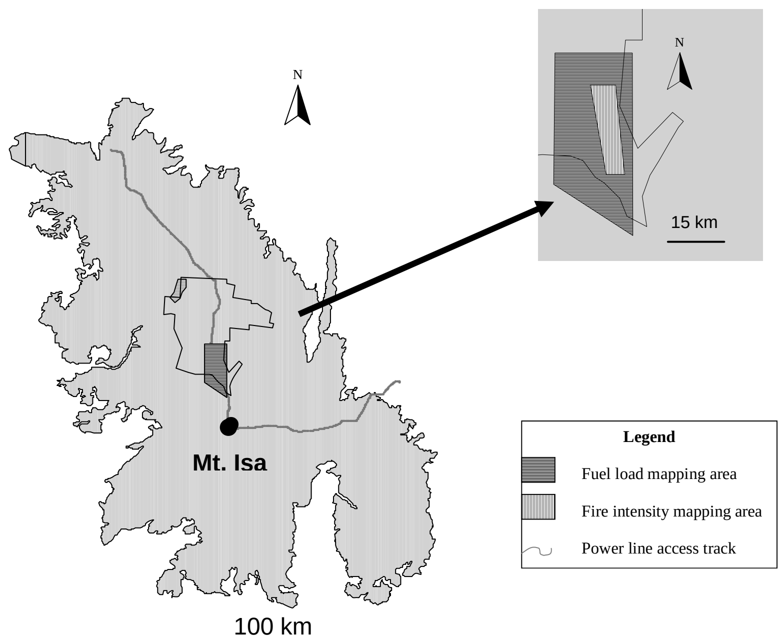

Figure 4.2 Location of the study area in the Mount Isa Inlier, Queensland, Australia. IKONOS imagery was used for mapping fuel load (horizontal shading) and fire intensity (right inset, vertical shading).

A fire escaped from a bush camp on Paroo Creek, east of the main study area, in November 2004 (Peter McNeven, pers. comm. 2005). The fire burnt westward and self-extinguished after number of days. Fortuitously, it covered some of the area where fuel load mapping was being undertaken. This provided an opportunity to investigate remote sensing of fire severity using fine scale IKONOS imagery. Potentially, the pre-fire fuel load map could be compared with a post-fire severity map. IKONOS satellite data were purchased for an area of approximately 100 km^2 centred on the power line (Figure 4.2 inset). This ensured good access to the fire scar. The cost of imagery prohibited data capture over a larger area.

4.2.2 Satellite Imagery

The IKONOS satellite sensor was chosen because the ensuing image pixel resolution is the finest scale commercially available. Resolution of IKONOS imagery ranges from 1 m for panchromatic (black and white) to 4 m for multi-spectral (colour), with data on blue, green, red and near infra-red (NIR) bands. This scale was important for investigating landscape heterogeneity. Attributes of the pre- and post-fire images are summarised in Table 4.1. Six separate images were used to produce the fuel load map due to file sizes and the computer processing power required.

Table 4.1 Attributes of the pre- and post-fire IKONOS images used to investigate the mapping of fuel loads and fire severity in the Mount Isa area.

Image	Capture date	Sun elevation (degrees)	Sun azimuth (degrees)	Datum and projection
Calton Hills (south)	18/7/2004	42.1560	31.970	WGS84 UTM S 54
Gunpowder (north)	18/7/2004	42.5820	32.4040	WGS84 UTM S 54
Post -fire area	2/4/2005	55.2450	46.2950	WGS84 UTM S 54

Image pre-processing

The images were purchased with a positional accuracy of 15 m, exclusive of terrain effects. This gave assurance that the area of interest had been captured but meant additional processing was required before analyses could be undertaken. The purpose of additional processing was to reduce potential errors related to geo-referencing, topography and atmospheric scattering. Major processing steps are outlined in Figure 4.3. Key points were (i) adjusting for the correct location on the Earth's surface (rectification); (ii) correcting for shadows due to topography and the position of the sun (cosine correction); (iii) correcting to reduce the effects of scattering due to dust and moisture in the atmosphere (atmospheric correction), and (iv) histogram matching so the bands on each image were adjusted to the same dynamic range. This final step provided for a seamless map upon completion. The following steps were undertaken using the software package ER Mapper (ERM 2003a):

1. **Image rectification**: A digital contour map of 1:100,000 scale with 10-m contour intervals was used for height information. These data were obtained from GeoScience Australia and used to make a Digital Elevation Model (DEM) for each area. The DEMs had a 10-m x 10-m resolution. The ER Mapper extension "Orthowarp" (ERM 2003b) was then used to rectify each of the 4 southerly images, based on the height models and 1:100,000 drainage and road data (also from GeoScience Australia). The two northern images were rectified to a 1:100,000 topographic map rather than a separate drainage layer. This topographic map was provided in jpeg file format and also had to be rectified before use. To rectify the images, visually distinct features such as creek junctions and road crossings were matched between the imagery and digital map data. After 10 - 12 well spread 'matches' were obtained, the best combination of 6 - 7 ground control points (GCP) was used for rectification. The Root Mean Square (RMS) error for each GCP was < 5 m.
2. **Cosine correction**: Differential brightness due to terrain illumination needs to be removed or minimised for accurate quantitative analysis (Wu *et al.* 2004). Law and

Nicol (2004) found that 'normalisation' worked best for IKONOS imagery. This method was applied initially but required significant computing capacity and was very time consuming. The 'cosine correction' method (Law & Nichol 2004) was then tested on a small, hilly portion of the image. It gave visually better results than normalisation so was applied to all images (Appendix 2). Information on slope, aspect and illumination was derived from the DEMs, and sun angle and elevation data were obtained from image metadata.

3. **Atmospheric correction**: Standard atmospheric correction procedures were used (Graetz & Gentle 1982). For each image, reflectance on the blue, green and red bands was graphed against NIR. For each band, the intercept of the 'soil line' with the y axis (i.e. when NIR = 0) was recorded. The average of the 6 intercept values for each band was used as the digital number correction. Digital numbers were adjusted on each image by deducting the relevant amount (blue 186.8, green 158.5, red 105.3). Note that the dynamic range of the image data was 11 bit (2,048 grey levels).
4. **Histogram matching**: Using one image as a standard, the histograms from each corresponding band of the other images were adjusted to the standard. This gave the same spread of values on all images upon completion.
5. **Compile final image**: Final images were made for each of the 6 areas after reflectance values were adjusted for shadowing and atmospheric interference, and histograms were matched.

The same pre-processing steps were undertaken for the post-fire image, except histogram matching was not necessary, since there was only one image.

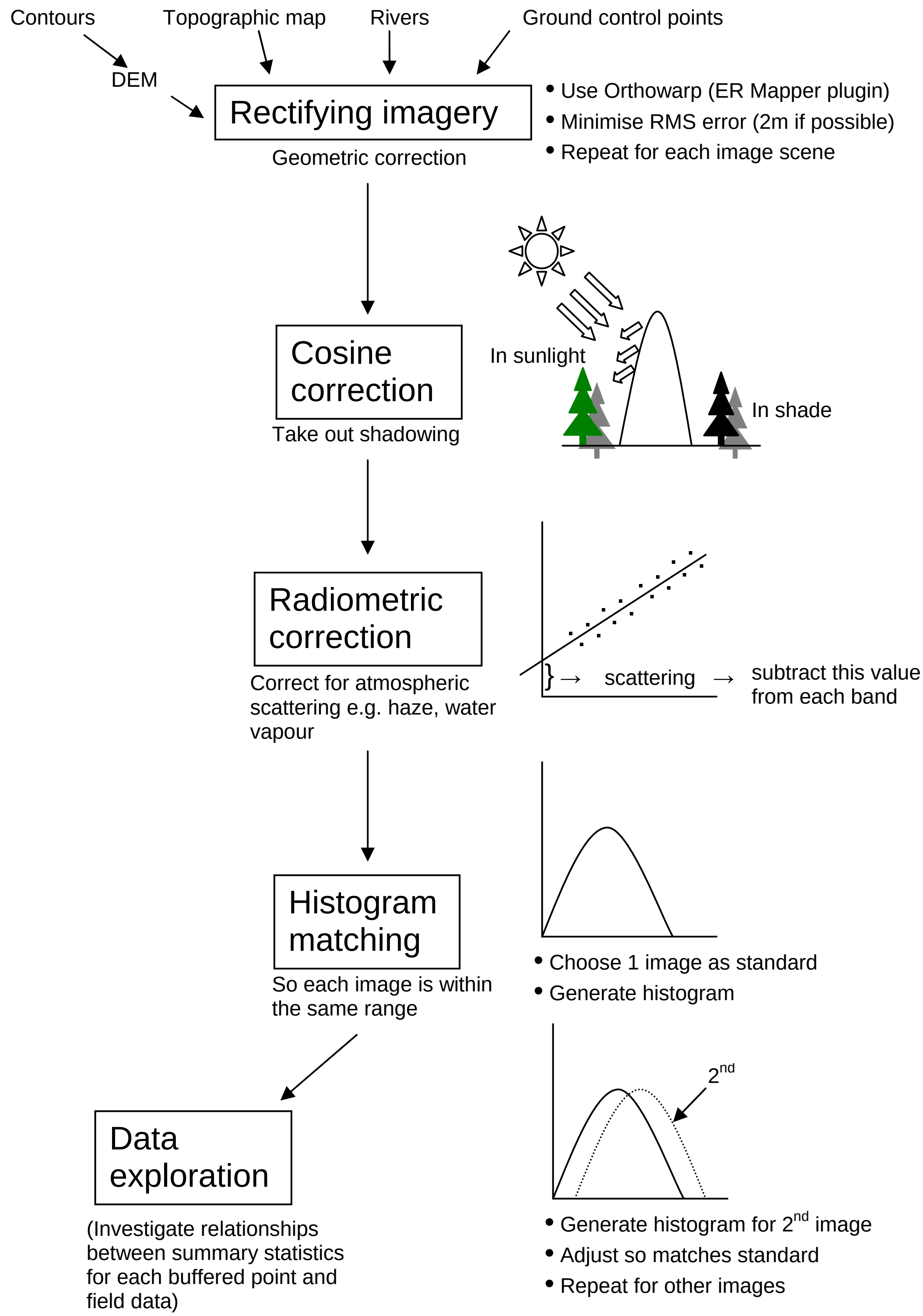

Figure 4.3 Summary of image processing steps undertaken to reduce errors related to geo-referencing, topography and atmospheric scattering.

4.2.3 Fuel Load Mapping

Fuel load measures in the field

Two pairs of field workers made fuel load estimates at 20 locations across the study area (Figure 4.4). Locations were based on vegetation type (with emphasis on spinifex/snappy gum communities), access, geographic spread, and fire age (to ensure fuel was estimated at a range of fire-age classes). At each location, data were recorded for 2 parallel transects, approximately 25 m apart. The starting point of the first transect was determined prior to field work using a GIS, which ensured its position was not biased by prior inspection. Subsequent transects were positioned by pacing approximately 25 m due south from the first transect starting point. Each transect was 500 m long and generally orientated east-west, or perpendicular to the access track to avoid crossing areas disturbed by roads. At 10 locations, data were recorded for 4 parallel transects; at the end of the first transect, each pair of recorders moved 50 m south and commenced a new transect on the return journey to the vehicle.

Data were recorded by walking along the transect on a pre-determined compass bearing and stopping every 25 m to record the location with a hand-held GPS (Garmin GPSMAP 76) and estimate fuel load. For efficiency, the interval between points was determined using a lightweight 25 m rope. The recorder held one end while the fuel-measurer walked to the full extent of the rope along the bearing, where the next GPS point was recorded and fuel measurements made. Fuel load estimates were made using the disc-drop method described in Chapter 3. The depth of fuel was measured at each point, walking in a consistent pattern each time (Figure 4.5). This gave 9 estimates of fuel load at approximately 1 disc-drop per pixel on a 3 x 3 cell grid. The dominant ground cover was recorded as spinifex (SP), other grass (OG) or both (SPOG). This cross-country method allowed for genuine landscape sampling. No allowances were made for staying within a particular vegetation type, soil type, topographic position or fire-age class. In total, fuel load data were obtained for 1,200 points across the study area.

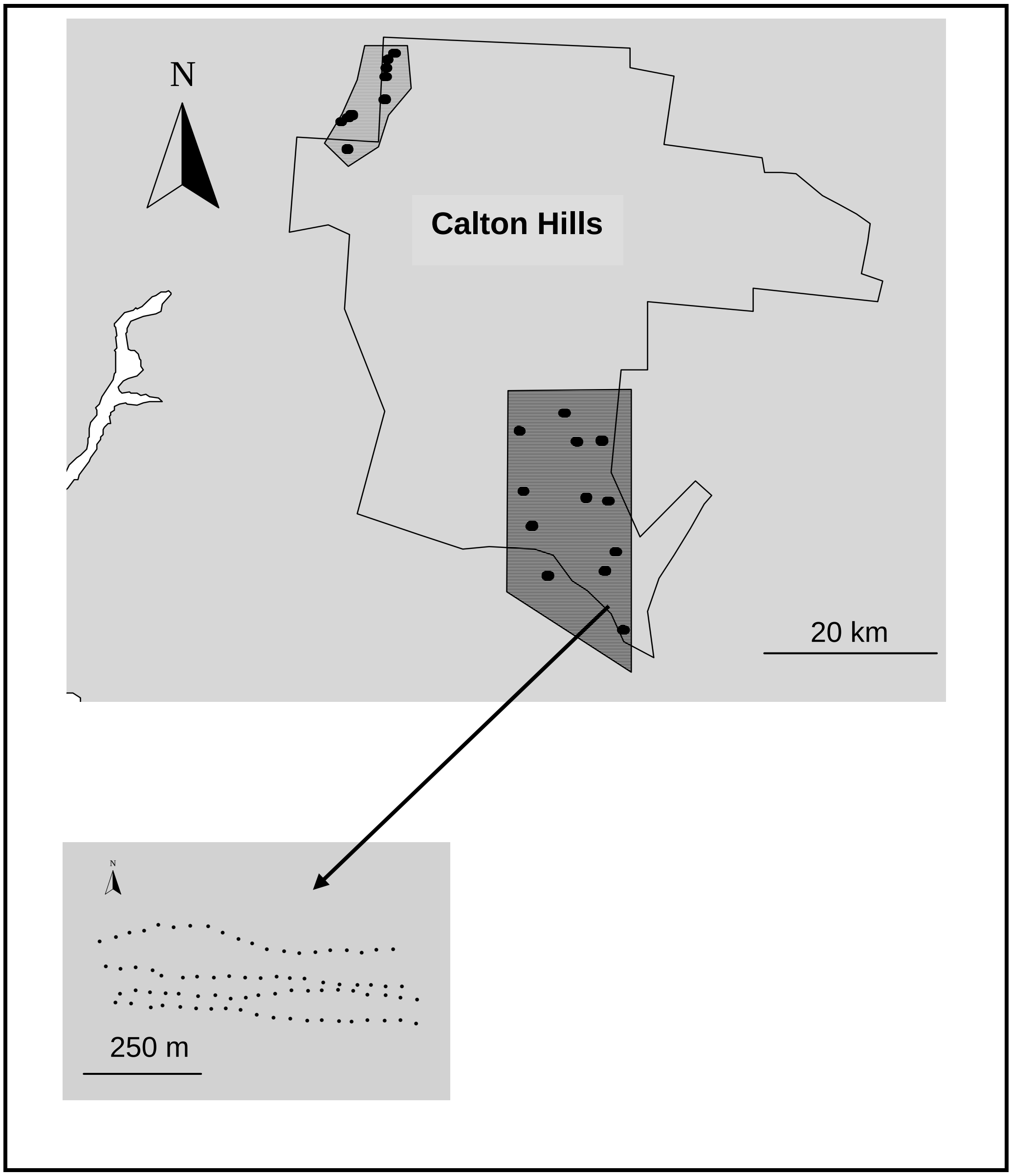

Figure 4.4 Fuel load estimates were made at 20 locations across the study area, 8 in the north and 12 in the south (black dots, main map). Data were recorded for two or four transects at each location, with 20 data points per transect (see inset).

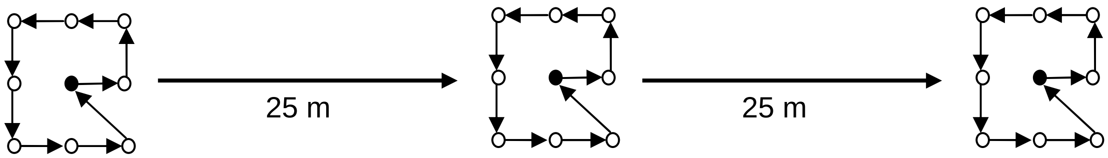

Figure 4.5 9 disc-drop measures were made at each transect point (black dot) to provide a fuel load estimate for a 3 x 3 pixel grid.

The cover of trees and shrubs

The cover of eucalypt trees was estimated using a basal area wedge. A circular sweep of the tree layer was made every 100 m. Trees were scored 1 or ½ depending on the distortion of their trunk (at breast height) by the optical wedge (Brack & Wood 1998). Tree species were recorded as snappy gum (*E. leucophloia*) or 'other'. Wedges of two different dimensions were used, based on availability. To ensure consistency, one sampling team used the 0.4 µm wedge while the other pair used the 0.7 µm wedge. Choosing a prism size for basal area measurement depends on the expected trunk thickness, with smaller factors suitable in sparse vegetation. Hassett *et al.* (2000) used a 0.5 µm wedge for sparse vegetation in their cross-Queensland study and a 2.0 µm wedge in rainforest. The 0.7 µm wedge used here gave a larger plot radius than the smaller wedge, and was at the upper end of the acceptable range for this vegetation (Rob Hassett pers. comm. 2003); however, it was considered acceptable given that wedge size was factored into the mathematical equation to convert field counts to basal area estimates.

The cover of shrubs was estimated at each transect point by assigning the nearest shrub in each quadrant to a height class. The species was recorded if known; otherwise 'other shrub' was noted. Height classes were estimated using a 2-m staff marked in centimetres, with each 25 cm segment numbered sequentially and marked with coloured tape for rapidly identifying the height class boundary. To calibrate height class against canopy area, height class and canopy dimensions were recorded for 98 specimens of *Acacia chisholmii*, the dominant shrub species in the area. A growth curve was produced from these data.

Data analysis

The average fuel depth at each point was converted to tones per hectare (t/ha) using the equation presented in Chapter 3, where disc-drop height was calibrated against fuel load. *Acacia* canopy area was estimated from the growth curve, and tree cover was calculated using the standard equation for its determination from basal area measurements (Brack & Wood 1998). Additional data fields were added at the site level as follows.

Fire-age data were determined from the fire scar map prepared in Chapter 2, with the fire age categories <1 year, 1 year, 2 years, 3 years, 4 years, and over 6 years. No transects were located on areas that had burnt 5 or 6 years ago. There were some issues to do with fire scar boundary placement (as discussed in Chapter 2), but fire-age data were readily adjusted based on field notes. For example, transect 19c appeared to be long unburnt (> 6 years) on the fire history map, but upon inspection half of the transect had been burnt in a 2001 fire. This could be determined by the size of spinifex clumps and distinct boundary between old and young spinifex. Similarly, transect 6 was on a small fire scar that had burnt within the previous 12 months and subsequent to production of the fire scar map.

Vegetation types were determined from a draft of the 1:100,000 regional ecosystem map being produced by the Queensland Environmental Protection Agency (Dan Kelman pers. comm. 2003) and from examining patterns apparent on the satellite imagery in conjunction with field notes. Vegetation was categorised into 5 broad types based on the dominant tree species: 1. *E. leucophloia* (Snappy Gum), 2. *E. pruinosa*, 3. *E. leucophylla* (Cloncurry Box), 4. *Acacia cambadgi* (Gidgee), 5. Mixed or other (e.g. Bloodwood, River Red Gum, White wood).

Landscape position categories included gullies, flats, footslopes, hillsides, ridges/crests, and bare ground (rocky outcrops, road edge, erosion areas, or areas with no vegetation) (Gunn *et al.* 1988). These were determined from the 10-m contour map and field notes. Points on drainage lines were determined from field notes, as well as a GIS exercise where a 5 m buffer was placed around the 1:100,000 drainage data. Points falling within the buffered area were designated as ‘drainage’.

Using the GIS software ArcView™ 3.3 (Environmental Systems Research Institute, Redlands, USA), the position of each fuel load point was plotted on a map of the study area. A 6 m buffer was created around each point to define a polygon that corresponded to the areas where fuel load estimates were made. This provided a file with uniquely identified buffered points, representing approximately 9 pixels, which was imported into ER Mapper. Statistics were obtained from the remotely sensed data for each buffered point or ‘local region’. Primarily, these were the mean and standard deviation of reflectance values for each of the four bands (blue, green, red and NIR). Location data were missing for one point, leaving a total of 1,199 points with corresponding field and reflectance measures.

A range of vegetation indices was calculated for each point because different indices perform better in different situations (Ray 1994) and their application to IKONOS imagery was uncertain. The remotely sensed data were used to calculate the Normalised Difference Vegetation Index (NDVI), NIR/Red (B4/B3), the Perpendicular Vegetation Index (PVI), an alternative Perpendicular Vegetation Index (PVI2) and the Enhanced Vegetation Index (EVI). Equations and references for these indices are given below. Digital numbers were used for all but the EVI calculation, where actual reflectance values were required (Kamel Didan, Terrestrial Biophysics and Remote Sensing Lab, University of Arizona, pers. comm. to Tina Lawson, JCU). For the EVI it was necessary to find the radiance and true reflectance value before the equation could be applied.

The statistical package SPSS (SPSS 2003) was used to test data for normality. Measures of fuel load, basal area and B4/B3 values were transformed (Log_{10} transformation) to improve normality. The data for *Acacia* canopy area did not conform to a normal distribution and could not be transformed, so non-parametric tests were used in analyses involving this variable.

Equations

NDVI

$\mathbf{NDVI = (\lambda_{NIR} - \lambda_{red}) / (\lambda_{NIR} + \lambda_{red})}$, where

λ_{NIR} and λ_{red} is the reflectance in the near infra-red and red bands respectively (Ray 1994).

B4/B3

$\mathbf{B4/B3 = \lambda_{NIR} / \lambda_{red}}$, where

λ_{NIR} and λ_{red} is the reflectance in the near infra-red and red bands respectively (Harris *et al* 2006).

PVI

$\mathbf{PVI = \sin(a)\, NIR - \cos(a)\, red}$, where

a = the angle between the soil line and the y-axis when NIR is plotted against red (Ray 1994).

PVI2

$PVI2 = [(NIR – a) * (red + b)] / Sqrt (1 + a^2)$, where

a = the intercept on the y-axis when NIR is plotted against red,

b = the slope of the soil line (Harris *et al.* 2006).

EVI

$EVI = G * (\rho_{NIR} – \rho_{Red}) / (\rho_{NIR} + C_1 * \rho_{Red} - C_2 * \rho_{Blue} + L)$, where

ρ_{NIR} = NIR reflectance

ρ_{Red} = Red reflectance

ρ_{Blue} = Blue reflectance

C_1 = Atmospheric resistance red correction coefficient (value = 6.0);

C_2 = Atmospheric resistance blue correction coefficient (value = 7.5);

L = Canopy background brightness correction factor (value = 1.0);

G = Gain factor (value = 2.5) (TBRS 2003).

The relationships between field measurements and reflectance values were explored using each band and index. Fuel load and cover increase with time since fire in spinifex grasslands (Chapter 3) and reflectance values were expected to change similarly. Determining the relationship between reflectance and fuel load would enable production of a fuel load map. Fire age classes were not equally represented, however. There were limited classes present on the study area and some were restricted in size so sample transects could not be equally allocated.

Statistical tests were undertaken based on the entire dataset and for data stratified for fire age (time since fire), grass type, tree type and landscape position. One-way analysis of variance was used to compare the distribution of data between different stratification categories and determine whether the means of each sample population were significantly different from one another. Tukey's 'honestly significantly difference' (HSD) test (SPSS 2003) was used to make pair-wise comparisons between the means and determine which pairs were significantly different. Correlations between site variables and reflectance values were investigated using Pearson's correlation coefficient if the data's distribution was normal, otherwise Spearman's Rank correlation coefficient was used.

The detected relationships were used to inform stepwise multiple regressions to identify a model for predicting fuel load from satellite derived data. For these analyses, the dependent variable was Log_{10} fuel load and the independent variables were the various bands and indices showing the strongest relationship with fuel load. A straight multiple regression was problematic because all bands and indices were correlated with one another (Table 4.2). Therefore, multivariate analyses were used to explore options for the best combination of variables. Multivariate analysis requires continuous data, not categorical data (Quinn & Keough 2002), so fire age and landscape position scores were converted to 'dummy variables' for these analyses. For example, new variables for fire age were 'fire age 0', 'fire age 1', 'fire age 2' and so on, with points scored '1' or '0' depending on whether they fell in that particular age class. Landscape position variables were re-categorised into 3 groups because fuel load was not significantly different between some locations (see below). This reduced the number of variables being considered. Some data sets had different units of measurement, so all exploratory models used standardised coefficients to enable comparison. Different model options were explored to predict fuel loads for 'spinifex', 'spinifex/other grass' and 'other grass' using the band or index that correlated best with fuel load for that grass species.

It was considered prudent to work with one band or index for the final map product. NDVI was chosen as the most appropriate index because it was easily calculated and had the strongest relationship with fuel load. Also, the mean NDVI at spinifex points was significantly different to the mean NDVI at ‘other grass’ points ($F = 9.37$, $p < 0.001$, $n = 1199$), allowing these two groups to be differentiated. The mean NDVI value for ‘spinifex/other grass’ (i.e. mixed) points was intermediate to the two other means but not significantly different to either (Figure 4.6). However, there were only 49 ‘spinifex/other grass’ points, representing 4% of the entire sample. This may have been an artefact of sampling, where cross-country transects recorded data in the main vegetation types as well as in their ecotones. Grouping spinifex with ‘spinifex/other grass’ ensured that areas of highest fuel load were captured, since both these types had significantly higher fuel load than sites that only contained other grass.

Table 4.2 Correlations between fuel load, bands and vegetation indices. Numbers in bold show correlations that are significant at the 0.01 level or better (2-tailed).

		Blue_x	Green_x	Red_x	NIR_x	EVI	NDVI	PVI	PVI2	Log_{10} Fld	Log_{10}B4/B3
	Pearson Correlation	1	0.908	0.833	0.797	-0.930	-0.549	0.501	0.844	-0.160	-0.548
Blue_x	Sig. (2-tailed)		**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199
	Pearson Correlation	0.908	1	0.871	0.832	-0.876	-0.574	0.522	0.892	-0.203	-0.572
Green_x	Sig. (2-tailed)	**0.000**		**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199
	Pearson Correlation	0.833	0.871	1	0.847	-0.914	-0.782	0.415	0.963	-0.288	-0.777
Red_x	Sig. (2-tailed)	**0.000**	**0.000**		**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199
	Pearson Correlation	0.797	0.832	0.847	1	-0.700	-0.349	0.835	0.941	-0.201	-0.345
NIR_x	Sig. (2-tailed)	**0.000**	**0.000**	**0.000**		**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199
	Pearson Correlation	-0.930	-0.876	-0.914	-0.700	1	0.818	-0.251	-0.838	0.231	0.817
EVI	Sig. (2-tailed)	**0.000**	**0.000**	**0.000**	**0.000**		**0.000**	**0.000**	**0.000**	**0.000**	**0.000**
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199
	Pearson Correlation	-0.549	-0.574	-0.782	-0.349	0.818	1	0.213	-0.593	0.281	0.999
NDVI	Sig. (2-tailed)	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**		**0.000**	**0.000**	**0.000**	**0.000**
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199
	Pearson Correlation	0.501	0.522	0.415	0.835	-0.251	0.213	1	0.614	-0.045	0.214
PVI	Sig. (2-tailed)	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**		**0.000**	0.119	**0.000**
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199
	Pearson Correlation	0.844	0.892	0.963	0.941	-0.838	-0.593	0.614	1	-0.256	-0.587
PVI2	Sig. (2-tailed)	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**		**0.000**	**0.000**
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199
	Pearson Correlation	-0.160	-0.203	-0.288	-0.201	0.231	0.281	-0.045	-0.256	1	0.278
Log_{10} Fld	Sig. (2-tailed)	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	0.119	**0.000**		**0.000**
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199
	Pearson Correlation	-0.548	-0.572	-0.777	-0.345	0.817	0.999	0.214	-0.587	0.278	1
Log_{10}B4/B3	Sig. (2-tailed)	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	
	N	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199	1,199

From this, areas of high fire risk could be identified; areas of other grass typically had lower fuel loads and were considered a lower wildfire risk. To determine the cut-off value for differentiation between the two groups, the mean and standard deviation of NDVI was determined for each grass type. However, one standard deviation from the mean NDVI for spinifex gave a value less than the mean NDVI for other grass. Therefore, the midpoint between the means of 'other grass' and 'spinifex/other grass' was chosen for the cut-off value (NDVI = 0.264). NDVI values equal to or greater than 0.264 were assigned to 'spinifex' and values less than 0.264 were considered 'other grass'. The probability of a site being spinifex and not other grass was determined by calculating the area under the NDVI distribution curve where NDVI ≥ 0.264.

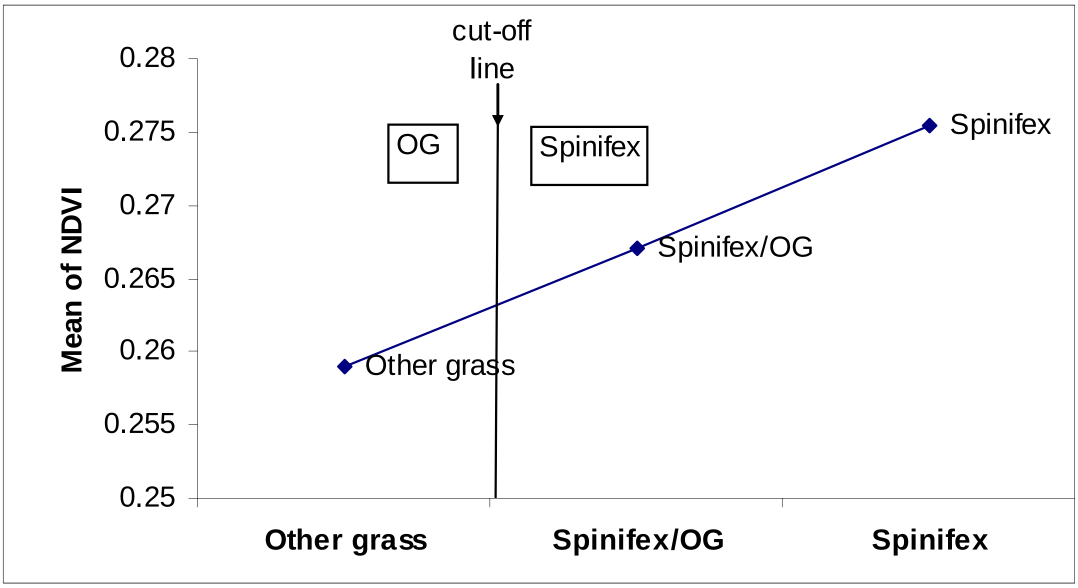

Figure 4.6 The mean NDVI for spinifex (n = 744) was significantly different to the mean NDVI of other grass, (n = 406), but not spinifex/other grass (n = 49). The mid-point value 0.264 was used to differentiate between 'spinifex' and 'other grass'.

To predict the fuel load of spinifex, the multi-linear regression model using NDVI and fire ages 0, 1, 2 and 3 gave the best result (Table 4.3). The following equation was then used for fuel load mapping:

$$\mathbf{Log_{10}FL_{(spin)} = 0.571 + 0.738*NDVI - 0.41*FA_{(0)} - 0.226*FA_{(1)} - 0.096*FA_{(2)} + 0.073*FA_{(3)}}$$

where FL is Fuel Load and FA is Fire Age at 0 (0 - 1 yr), 1 (1 - 2 yrs), 2 (2 - 3 yrs) and 3 (3 - 4 yrs). The predicted fuel load at each point was compared with field estimates of fuel load.

Table 4.3 Summary of 5 model options for predicting Log_{10} Fuel Load. For model 5, F = 57.4, p < 0.001, df = 793.

Model predictors	R	R^2	Adjusted R^2	Std. Error of the Estimate	Change Statistics				
					R^2 Change	F Change	df1	df2	Sig. F Change
1. (Constant) FireAge0	0.378	0.143	0.142	0.27912	0.143	132.149	1	792	0.000
2. (Constant) FireAge0, FireAge1	0.465	0.217	0.215	0.26704	0.074	74.312	1	791	0.000
3. (Constant) FireAge0, FireAge1, NDVI	0.494	0.244	0.241	0.26246	0.028	28.820	1	790	0.000
4. (Constant) FireAge0, FireAge1, NDVI, FireAge2	0.509	0.259	0.255	0.26000	0.015	16.012	1	789	0.000
5. (Constant) FireAge0, FireAge1, NDVI, FireAge2, FireAge3	0.517	0.267	0.262	0.25881	0.008	8.300	1	788	0.004

Fuel map compilation

A map of spinifex fuel load was prepared using ArcView 3.3 GIS software in conjunction with ER Mapper. A grid of digital data was prepared for each independent variable in the model. These grids were combined mathematically to produce the final fuel load map. The process is depicted in Appendix 3 and summarised as follows:

- The NDVI variable was prepared using ER Mapper and the standard NDVI formula. To select for spinifex, values ≥ 0.264 were retained and null values were assigned to remaining pixels. The spinifex cells were multiplied by the coefficient 0.738 to produce the NDVI variable to which the constant 0.571 was added, as per the linear regression equation. This file was saved for later import into ArcView.
- The fire history map prepared in Chapter 2 was reclassified to produce a grid surface for each fire age.
- A grid of Log_{10} fuel load was prepared in ArcView by combining the spinifex variable grid with the fire age grids.
- This process was repeated for each of the six image areas, with the final fuel load grids combined to produce a larger fuel load map.

The shrub layer's contribution to the fuel load was not modeled explicitly, but identifying areas of *Acacia* would allow some account to be given to this fuel source. Using a similar process to that described for spinifex, each data point was scored (1) for the presence of *Acacia*, (2) for the presence of another shrub genus, or (3) if shrubs were absent. Analysis of variance was used to determine whether the mean reflectance was significantly different between these three groups for each band and index. The intention was to identify areas with *Acacia*, obtain a realistic estimate of fuel load from the literature, and add this to the ground cover-derived fuel load map.

4.2.4 Fire Severity Mapping

Three approaches were investigated for fire severity mapping. First, correlations between fire severity field data and the reflectance values of different bands and indices were investigated. Strong relationships would allow a fire severity map to be derived from first principles, as described above for fuel load. Secondly, the change in NDVI between pre- and post-fire images was calculated and compared with field values. This method has been successfully applied by others using broader scales of imagery (Diaz-Delgado *et al.* 2003; Hammill & Bradstock 2006; Ruiz-Gallardo *et al.* 2004). Thirdly, fire intensity was inferred from the fuel load map using the equation **I = HwR**. The heat of combustion (H) was assumed to be constant for the spinifex ground cover; the weight of fuel consumed (w) was obtained from the fuel load map, assuming complete combustion; and rate of spread (R) was estimated from slope, on the basis that R doubles for every 10^0 increase in slope (Luke & McArthur 1978). Weather conditions were not taken into account. Slope values were derived from the DEM and variables were combined as shown in Appendix 3. Comparisons were then made between predicted intensity values and on-ground fire severity measures.

Field measures of fire severity

Chapter 3 (Section 3.2.3) describes the methods used to record fire severity. The data used for detecting variation in fire severity within a single fire scar also allowed for on-ground measurements to be compared with reflectance values on the post-fire image (Figure 4.7). Data on species, tree and shrub char heights, canopy scorch, shrub twig diameter, and resulting severity and patchiness scores were available for ten sites (500 points). These covered a range of representative vegetation types and landscape positions.

Data analysis

Data were processed as described in section 4.2.3 above, where fuel load measures were related to reflectance values. Using ArcView 3.3, each sample point was located on a map and linked with its corresponding severity score (see 3.2.3). Points were surrounded by a 6 m buffer to identify 'local-regions'. For each local region, statistical information was obtained for each band of the remotely sensed image using ER Mapper. A summary file was created to combine field values, image values and index values, as previously described (in 4.2.3). Data for twig diameter, band 1 (blue) reflectance, B4/B3 and NDVI were transformed (Log_{10}) to improve normality. The distribution of fire severity scores was skewed towards the right and could not be easily transformed. Instead of restricting analyses to non-parametric tests for this variable, severity scores were used as categorical data, with 10 categories defined to adjust for skewness. Statistical relationships between field values and reflectance values were investigated, with emphasis on tree char height. Each point represented a tree, so the data set for tree char height was complete. Also, tree char height was measured, rather than estimated, so using tree data was potentially most accurate.

Reflectance values were compared to field values for the entire dataset and then by stratifying data by tree species and shrub species. One-way analysis of variance was used to compare the distribution of data for each stratification category and Tukey's HSD test was used to determine which sample means were significantly different from one another. Correlations between site variables and reflectance values were investigated using Pearson's correlation coefficient or Spearman's Rank correlation coefficient depending on the distribution of data.

The established NDVI differencing method for estimating fire severity is: $NDVI_{difference} = NDVI_{pre\text{-}fire} - NDVI_{post\ fire}$ (see Hammill & Bradstock 2006), with larger $NDVI_{difference}$ values associated with areas of greater fire severity. $NDVI_{difference}$ values were calculated for each point and compared to on-ground fire severity measures using Spearman's Rank correlation coefficient.

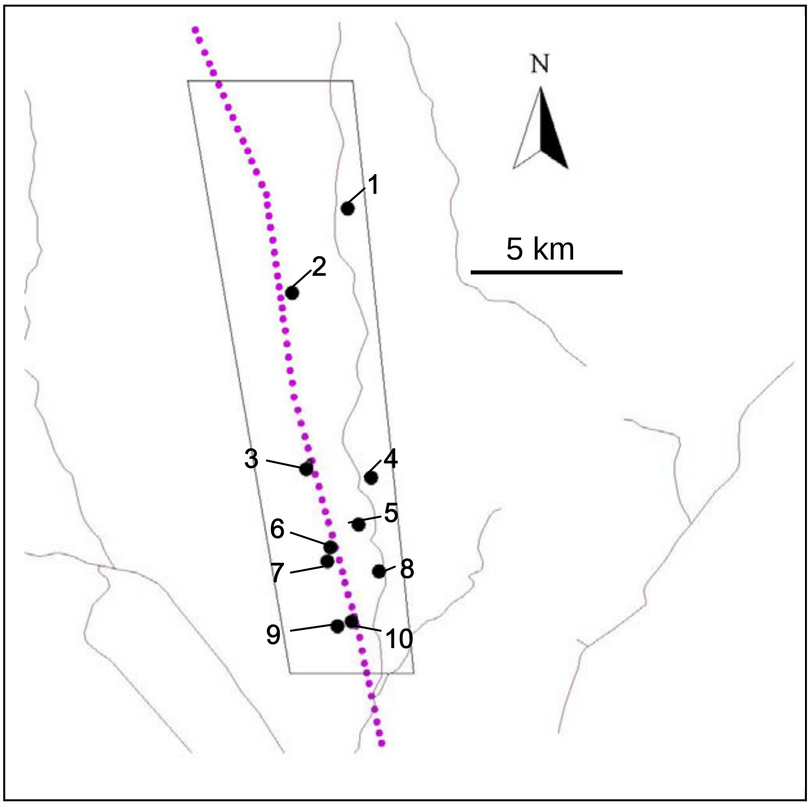

Figure 4.7 Ten locations (numbered), each with two transects where fire severity data were collected for relating to reflectance values on satellite imagery (area of interest is shown by rectangle). The powerline (purple dots) and road access tracks (grey lines) are also shown.

4.3 Results and Interpretations

4.3.1 The Pre-Fire Landscape

Fuel load, fire age and reflectance

Fuel load was related to fire age, increasing up to year 3 before levelling off (Figure 4.8). This growth curve is typical for fuel in savanna landscapes where fuel load levels off because rates of fuel production and decay become equal (Cook 2003), although these rates vary with rainfall, soil depth and fertility. There was a wide spread of values at each age class due to natural fuel discontinuity in hummock grasslands.

Reflectance was low soon after fire (black ground), but increased dramatically in year 1 before dropping to a moderate level thereafter (Figure 4.9). This relationship was expected to be a growth curve, with a rapid change initially as soil dominated the response but levelling off as grass hummocks grew and cover increased. The initial marked change in reflectance was likely due to rapid post-fire regeneration and the growth flush of ephemeral species. Subsequent decline in NIR reflectance was probably due to the hardening and partial senescence of spinifex clumps.

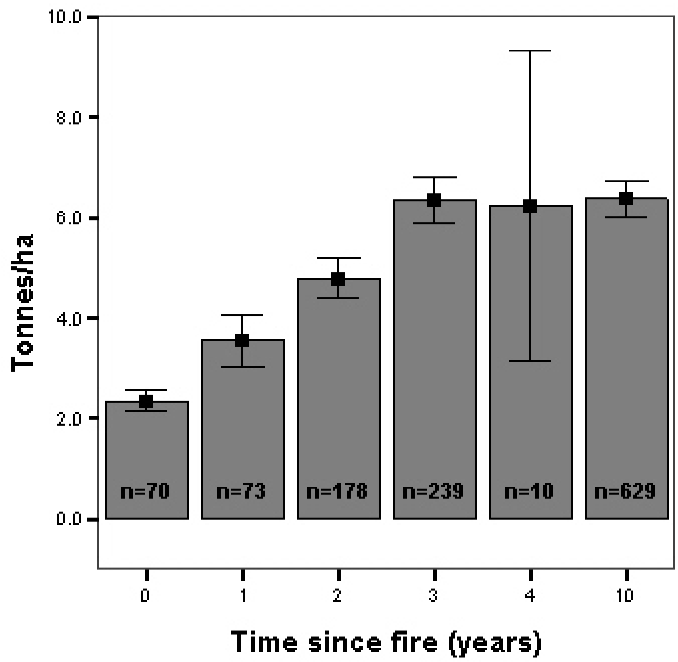

Figure 4.8 Fuel load increased with time since fire, but levelled off after three years. (n = number of points; Error bars show 95% confidence intervals).

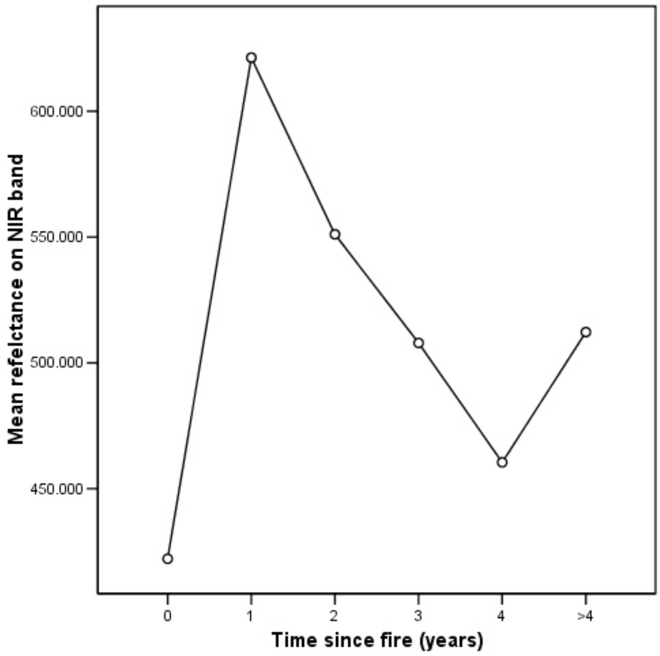

Figure 4.9 Mean reflectance was low after fire, increased in the immediate post-fire years, then dropped to a moderate level. This general pattern was apparent on all bands.

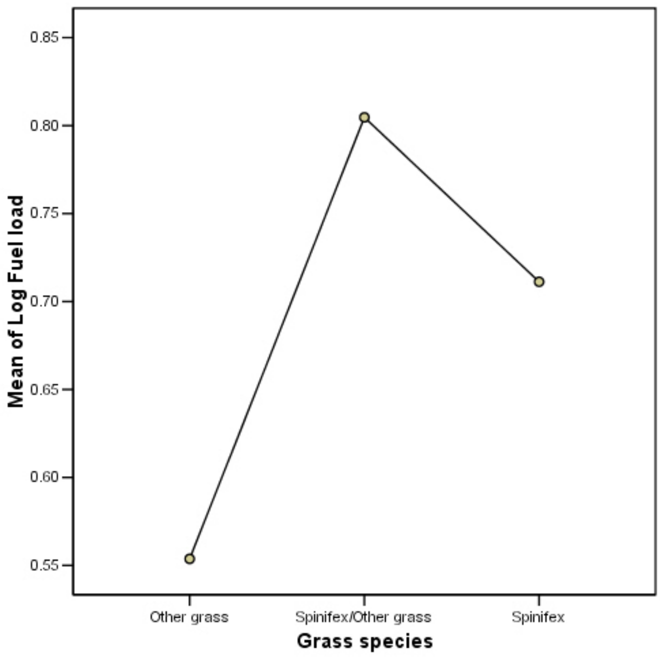

Figure 4.10 Fuel load was highest at sites with both spinifex and other grass, and lowest at sites with species other than spinifex (F = 49.6, p < 0.001, df = 2).

Table 4.4 Fuel load was weakly but significantly correlated with all bands and indices except PVI.

		Log_{10} Fld
Log_{10} Fld	Pearson Correlation	1
	Sig. (2-tailed)	
	N	1,199
Blue_x	Pearson Correlation	**-0.160**
	Sig. (2-tailed)	0.000
	N	1,199
Green_x	Pearson Correlation	**-0.203**
	Sig. (2-tailed)	0.000
	N	1,199
Red_x	Pearson Correlation	**-0.288**
	Sig. (2-tailed)	0.000
	N	1,199
NIR_x	Pearson Correlation	**-0.201**
	Sig. (2-tailed)	0.000
	N	1,199
EVI	Pearson Correlation	**0.231**
	Sig. (2-tailed)	0.000
	N	1,199
NDVI	Pearson Correlation	**0.281**
	Sig. (2-tailed)	0.000
	N	1,199
Log_{10} B4/B3	Pearson Correlation	**0.278**
	Sig. (2-tailed)	0.000
	N	1,199
PVI	Pearson Correlation	-0.045
	Sig. (2-tailed)	0.119
	N	1,199
PVI2	Pearson Correlation	**-0.256**
	Sig. (2-tailed)	0.000
	N	1,199

Bold: Correlation is significant at the 0.01 level (2-tailed).

Landscape position and fuel load

Landscape position affected fuel load (Figure 4.11) with bare areas/rocky outcrops having least fuel, followed by flats, gullies and foot-slopes. Fuel load was greatest on mid-slopes ($F = 19.88$, $p < 0.001$, $df = 5$). Grazing and grass type may account for these differences; other grass was more prevalent on flats, which were also heavily grazed. There was much overlap in the spread of values, however (Figure 4.12), suggesting that landscape position was secondary to fire age as driver of fuel load.

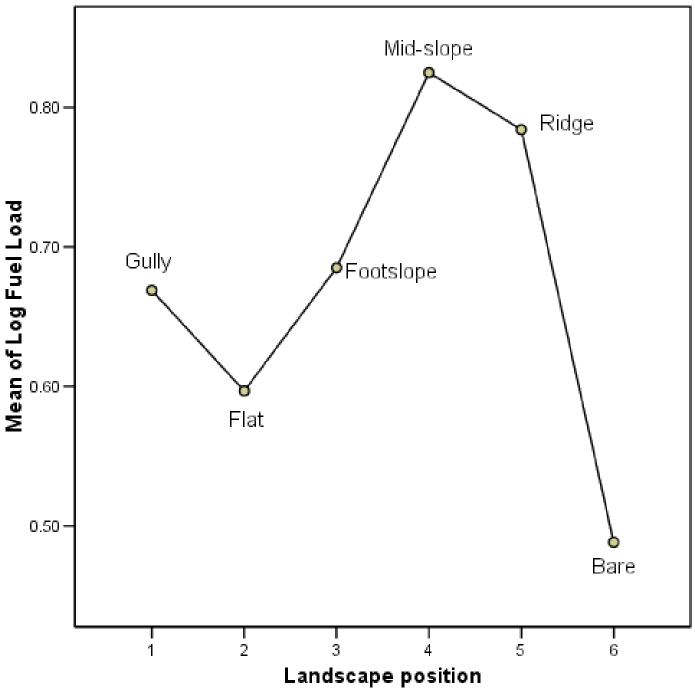

Figure 4.11 Mean fuel load varied with landscape position. Most fuel occurred on mid-slopes, whereas gullies, flats and footslopes had similar amounts of fuel.

Points stratified for landscape position showed that fuel load was positively correlated with reflectance on some bands, at some landscape positions. Although the relationships were significant, they were weak, and no single index was consistently more predictive than another. Therefore landscape position was not considered a strong variable for generating the fuel load map. Regarding proximity to drainage lines, there was no significant difference in fuel load between samples within or adjacent to drainage lines, and those more than 5 m from a drainage line. This was unexpected since available soil moisture has a strong influence on plant germination and growth in arid and semi-arid environments. Many points recorded as ‘drainage’, however, were bare areas in dry creek beds.

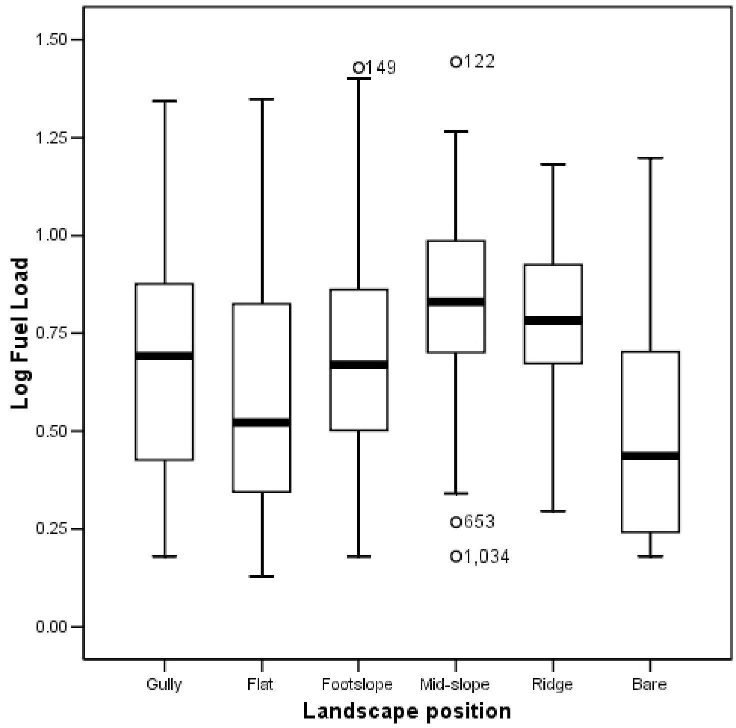

Figure 4.12 Mean fuel load was significantly different between some landscape positions; however, each category had a wide spread of values. Numbered points represent site IDs of outlying values.

Canopy dimensions of *Acacia chisholmii*

The growth curve for *Acacia chisholmii* is shown in Figure 4.13. There was a strong relationship between shrub height and canopy area ($R^2 = 0.8006$, $p < 0.001$, $n = 98$), so the model was considered suitable for estimating cover of *Acacia chisholmii*. This was calculated at each point based on the equation:

$$\mathbf{y = 56.73 \ln(x) + 152.2}$$

where y = shrub height (in cm) and x = canopy area (in m^2) ($F = 389.312$, $p < 0.001$, $n = 98$).

Acacia *dimensions, fuel load relationships and reflectance*

Acacia height increased with time since fire (Figure 4.14), and mean heights were significantly different when points were stratified for fire age ($F = 65.04$, $p < 0.001$, df = 5). There was a concomitant increase in canopy area. Both height and canopy area were weakly but significantly correlated with fuel load (***Acacia* height**: Pearson's $R = 0.419$, $p < 0.001$, $n = 283$; ***Acacia* canopy area**: Spearman's $R = 0.37$, $p < 0.001$, $n = 283$), but fuel load itself increased with time since fire. There was therefore an issue of collinearity, and since not all areas had *Acacia*, fuel load was mapped using ground cover alone. A more accurate result was anticipated by identifying areas with *Acacia* and separately accounting for its contribution to the fuel load.

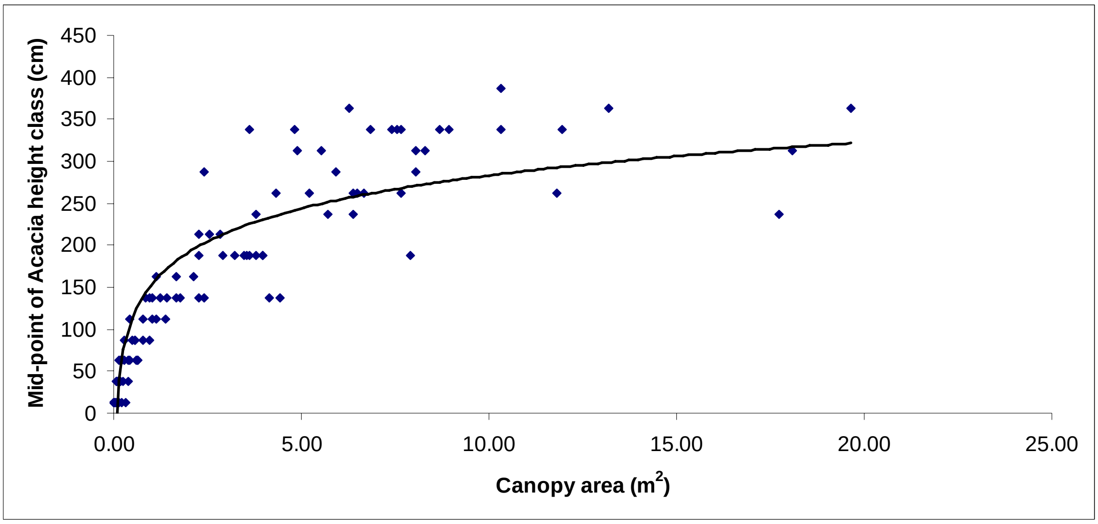

Figure 4.13 The canopy area of *Acacia chishomii* increased with *Acacia* height ($y = 56.73 \ln(x) + 152.2$; $R^2 = 0.8006$, $p < 0.001$, n=98).

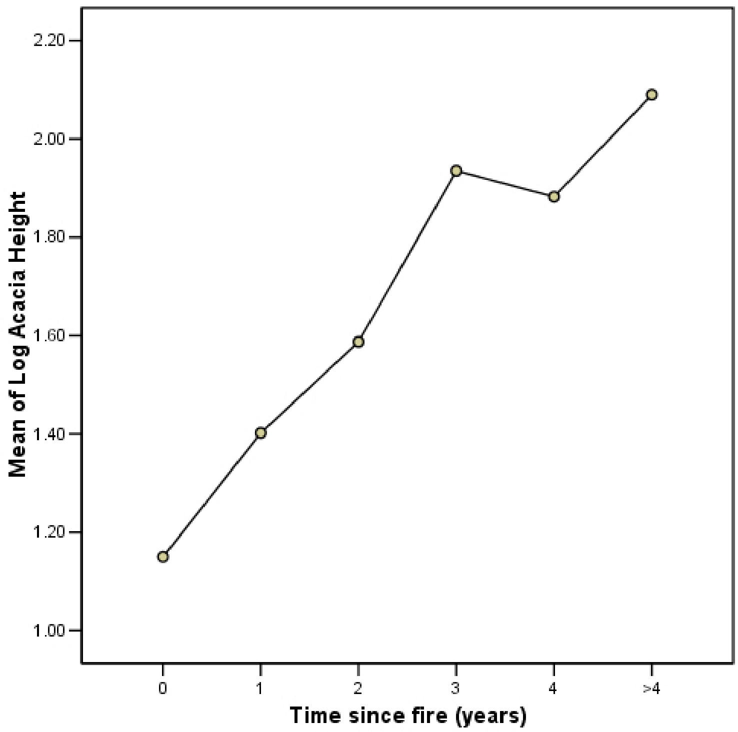

Figure 4.14 The average height of *Acacia chisholmii* increased with time since fire. (*Acacia* height was measured in cm).

Reflectance values would potentially relate more strongly to canopy area than *Acacia* height, given the aerial view provided by satellite imagery. The distribution of height measurements could be normalised using a simple transformation (Log_{10}), however, so both variables were used in exploratory analysis. *Acacia* height was correlated with some bands and indices (Table 4.5) as was *Acacia* canopy area (Table 4.6). *Acacia* canopy area was negatively correlated with reflectance on the green, red and NIR bands.

Table 4.5 The height of *Acacia* was correlated to reflectance on the red band as well as EVI, NDVI, PVI2, and log B4/B3.

		Log_{10} B4/B3
Log_{10} B4/B3	Pearson Correlation	1
	Sig. (2-tailed)	
	N	283
Log_{10} Fld	Pearson Correlation	**0.419**
	Sig. (2-tailed)	0.000
	N	283
Blue_x	Pearson Correlation	-0.026
	Sig. (2-tailed)	0.663
	N	283
Green_x	Pearson Correlation	-0.093
	Sig. (2-tailed)	0.117
	N	283
Red_x	Pearson Correlation	**-0.213**
	Sig. (2-tailed)	0.000
	N	283
NIR_x	Pearson Correlation	-0.067
	Sig. (2-tailed)	0.264
	N	283
EVI	Pearson Correlation	**0.159**
	Sig. (2-tailed)	0.007
	N	283
NDVI	Pearson Correlation	**0.277**
	Sig. (2-tailed)	0.000
	N	283
PVI	Pearson Correlation	0.094
	Sig. (2-tailed)	0.113
	N	283
PVI2	Pearson Correlation	**-0.170**
	Sig. (2-tailed)	0.004
	N	283
Log_{10} B4/B3	Pearson Correlation	**0.277**
	Sig. (2-tailed)	0.000
	N	283

**. Correlation is significant at the 0.01 level (2-tailed).

Table 4.6 The canopy area of *Acacia* was negatively correlated to reflectance on the green, red and NIR bands.

		***Acacia* canopy area**
***Acacia* canopy area**	Correlation Coefficient	1.000
	Sig. (2-tailed)	
	N	284
Log_{10} Fld	Correlation Coefficient	**0.370**
	Sig. (2-tailed)	0.000
	N	284
Blue_x	Correlation Coefficient	*-0.137*
	Sig. (2-tailed)	0.021
	N	284
Green_x	Correlation Coefficient	**-0.225**
	Sig. (2-tailed)	0.000
	N	284
Red_x	Correlation Coefficient	**-0.321**
	Sig. (2-tailed)	0.000
	N	284
NIR_x	Correlation Coefficient	*-0.136*
	Sig. (2-tailed)	0.022
	N	284
EVI	Correlation Coefficient	**0.281**
	Sig. (2-tailed)	0.000
	N	284
NDVI	Correlation Coefficient	**0.358**
	Sig. (2-tailed)	0.000
	N	284
PVI	Correlation Coefficient	0.086
	Sig. (2-tailed)	0.149
	N	284
PVI2	Correlation Coefficient	**-0.247**
	Sig. (2-tailed)	0.000
	N	284
Log_{10} B4/B3	Correlation Coefficient	**0.358**
	Sig. (2-tailed)	0.000
	N	284

Bold. Spearman's rho correlation is significant at the 0.01 level (2-tailed).
Italic. Spearman's rho correlation is significant at the 0.05 level (2-tailed).

The tree layer, fuel load relationships and reflectance

There was no significant difference between mean tree cover and fire age at the stand level. When separated into species, basal area did not change significantly with fire age for snappy gum; however, mixed stands showed slight but significant differences in cover over time ($F = 2.9$, $p < 0.02$, $n = 194$, $df = 5$). This response can be accounted for by the fire response of eucalypt trees. Resprouting from the trunk would not affect basal area estimates, and resprouting from the base to give multi-stemmed specimens potentially resulted in basal area estimates similar to larger, single stems before a fire. Tree cover was weakly correlated with some bands of reflectance (Table 4.7), but not when stratified into species. This suggests that there is an overall relationship between reflectance and tree cover, but not the cover of individual tree species.

The fuel load map was developed based on ground cover alone. Although mean fuel load differed between vegetation types (ANOVA $F = 21.8$, $p < 0.001$, $n = 1,198$, $df = 4$), differences were slight and the sample design may not have adequately tested this relationship. Any differences may have been a manifestation of landscape position*; E. leucophylla* and *E. pruinosa* were more common on flats, and flats had lower fuel load than mid-slopes where *E. leucophloia* was more common

Models to predict fuel load

For spinifex, spinifex/other grass and other grass alone, fire age and the discriminating band or index (NDVI, Green and PVI2 respectively) were the strongest indicators of fuel load. Including fire age classes beyond 3 years did not improve predictive power, because mean fuel loads were not significantly different at greater fire ages. Models were not improved by including landscape position as an additional independent variable.

The model for spinifex had the strongest predictive power ($R^2 = 0.279$, $p < 0.001$). When spinifex and spinifex/other grass sites were combined, the predictive power fell by only 1.5% ($R^2 = 0.264$, $p < 0.001$). This model was used for fuel load mapping because spinifex/other grass sites had the highest fuel load and spinifex occurred at the majority of sites. The equation using both grass groups accounted for the greatest proportion of the landscape as well as areas of greatest fire hazard.

When possible NDVI values were substituted into the model to estimate potential fuel load at different fire ages, predictions (approximately 2 - 15 t/ha) were within the range obtained for field measurements. However, fuel load could be over-estimated at recently burnt sites and under-estimated at long-unburnt sites. This was due to the cut-off value assigned to NDVI for spinifex (0.264) and the range of NDVI values present on the image. When NDVI was 0.264 or greater, there was a sixty percent chance that the site contained spinifex. However, it meant that the minimum fuel load depicted by the map would be 2.3 t/ha at recently burnt sites, although sample points may have been bare. Conversely, fuel load estimates were constrained by the largest NDVI value present on the image, in this case 0.715. This meant mapped fuel load values could not exceed 15 t/ha. Nonetheless the model was considered suitable. The resulting map would show areas of high, medium and low fuel, thus enabling relative comparisons to be made between areas, rather than precise fuel load estimates.

Table 4.7 Tree cover was weakly correlated with some bands of reflectance; however, there were no significant correlations when 'Snappy Gum' and '|Other tree species' were considered separately.

		Log_{10} Stand Basal Area	Log_{10} Snappy Gum Basal Area	Log_{10} Other Tree Basal Area
Blue_x	Correlation Coefficient	*-0.129*	-0.073	-0.044
	Sig. (2-tailed)	0.037	0.336	0.532
	N	264	177	200
Green_x	Correlation Coefficient	**-0.160**	-0.082	-0.078
	Sig. (2-tailed)	0.009	0.278	0.272
	N	264	177	200
Red_x	Correlation Coefficient	**-0.173**	-0.087	-0.106
	Sig. (2-tailed)	0.005	0.247	0.135
	N	264	177	200
NIR_x	Correlation Coefficient	-0.058	0.031	-0.080
	Sig. (2-tailed)	0.346	0.680	0.263
	N	264	177	200
EVI	Correlation Coefficient	**0.186**	0.133	0.065
	Sig. (2-tailed)	0.002	0.078	0.362
	N	264	177	200
NDVI	Correlation Coefficient	**0.194**	0.146	0.077
	Sig. (2-tailed)	0.001	0.053	0.278
	N	264	177	200
PVI	Correlation Coefficient	0.044	0.104	-0.014
	Sig. (2-tailed)	0.477	0.169	0.839
	N	264	177	200
PVI2	Correlation Coefficient	*-0.125*	-0.034	-0.097
	Sig. (2-tailed)	0.042	0.653	0.171
	N	264	177	200
Log_{10} B4/B3	Correlation Coefficient	**0.194**	0.146	0.077
	Sig. (2-tailed)	0.001	0.053	0.278
	N	264	177	200
Log_{10} St_BA	Correlation Coefficient	1.000	**0.737**	**0.590**
	Sig. (2-tailed)	.	0.000	0.000
	N	264	177	200
Log_{10} Sn_BA	Correlation Coefficient	**0.737**	1.000	-0.036
	Sig. (2-tailed)	0.000	.	0.701
	N	177	177	113
Log_{10} OT_BA	Correlation Coefficient	**0.590**	-0.036	1.000
	Sig. (2-tailed)	0.000	0.701	.
	N	200	113	200

Bold. Spearman's rho correlation is significant at the 0.01 level (2-tailed).
Italic. Spearman's rho correlation is significant at the 0.05 level (2-tailed).

The Acacia constant

Sites with *Acacia*, other shrub species or no shrubs had significantly different mean reflectance on all bands and for all indices (Table 4.8). For every case except NDVI, the mean reflectance for *Acacia* species was significantly lower than the mean reflectance at sites with no shrubs. Sites with other shrub species had greatest reflectance. This relationship was strongest for Band 2 (green) ($F = 54.747$, $p < 0.001$, $n = 499$), but there was considerable overlap in the spread of values (Figure 4.15). The mean digital number for *Acacia* on the green band was 209; at this point 60% of *Acacia* cases had been accounted for, there was an 18% chance that the area contained 'other shrubs' and a 22% chance that no shrub was present. *Acacia* sites were therefore isolated by selecting all values DN < 209 on the green band using ER Mapper. This was imported into ArcView 3.3 and converted to a grid for adding the *Acacia*'s contribution to fuel load as a constant (Appendix 3). The value of one tonne per hectare was arbitrarily assigned for *Acacia* fuel load – a more reliable estimate was not available in the literature. The assigned value was for indicative purposes only, to demonstrate how shrubs could be accounted for.

Table 4.8 The mean reflectance of sites stratified for '*Acacia*', 'other shrub species' or 'no shrubs' was significantly different on all bands and for all indices. The strongest difference was detected on the green band (ANOVA F = 54.747, p < 0.001, n = 499).

		Sum of Squares	df	Mean Square	F	Sig.
	Between Groups	38,659.349	2	19,329.674	42.919	0.000
Blue_x	Within Groups	538,649.875	1,196	450.376		
	Total	577,309.224	1,198			
	Between Groups	298,451.829	2	149,225.915	**54.747**	0.000
Green_x	Within Groups	3,259,966.776	1,196	2,725.725		
	Total	3,558,418.605	1,198			
	Between Groups	300,182.238	2	150,091.119	30.898	0.000
Red_x	Within Groups	5,809,740.729	1,196	4,857.643		
	Total	6,109,922.967	1,198			
	Between Groups	467,551.650	2	233,775.825	40.792	0.000
NIR_x	Within Groups	6,854,156.942	1,196	5,730.900		
	Total	7,321,708.592	1,198			
	Between Groups	2.514	2	1.257	32.288	0.000
EVI	Within Groups	46.557	1,196	0.039		
	Total	49.071	1,198			
	Between Groups	0.062	2	0.031	7.588	0.001
NDVI	Within Groups	4.883	1,196	0.004		
	Total	4.945	1,198			
	Between Groups	74,061.586	2	37,030.793	26.632	0.000
PVI	Within Groups	1,662,971.870	1,196	1,390.445		
	Total	1,737,033.456	1,198			
	Between Groups	13,446,716.379	2	6,723,358.189	36.447	0.000
PVI2	Within Groups	220,626,450.530	1,196	184,470.276		
	Total	234,073,166.908	1,198			
	Between Groups	0.057	2	0.029	7.656	0.000
Log_{10} B4/B3	Within Groups	4.455	1,196	0.004		
	Total	4.512	1,198			

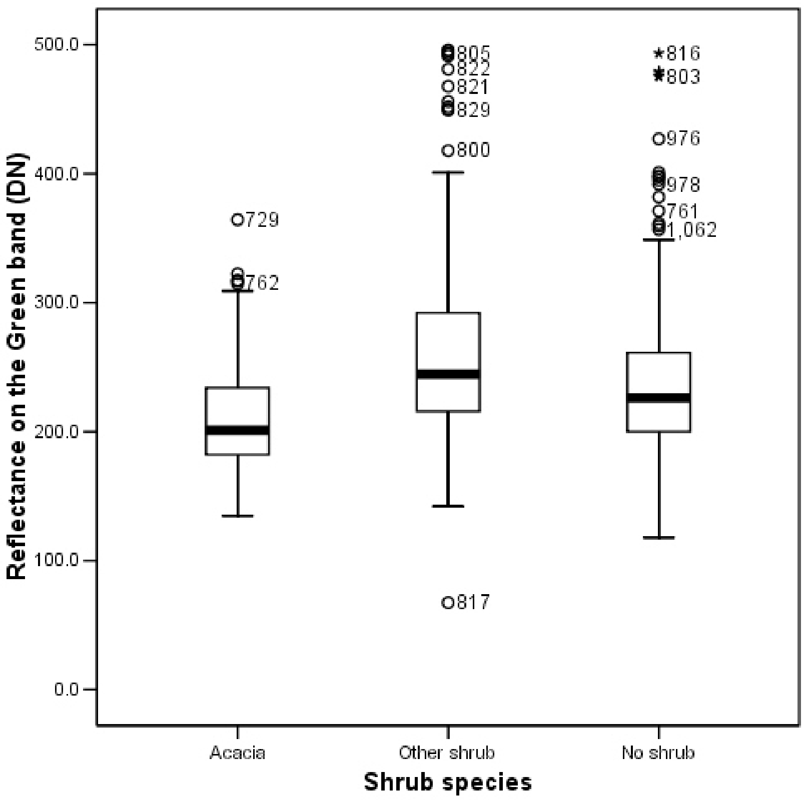

Figure 4.15 Although mean reflectance was significantly different between shrub species, there was considerable overlap in values. Numbered points represent IDs of outlying values.

Fuel map compilation and model predictions

A visual assessment of the fuel load maps prepared for the two study areas at Calton Hills station (Figure 4.16 and Figure 4.17) suggests that fuel load is lower on the flat riverine plains, in keeping with the statistical analysis presented above. However, the apparently higher fuel load along drainage lines was not evident from field measures. This can be explained by the inclusion of NDVI in the fuel load model. This index is used to measure the biomass of plants, which is typically high along vegetated drainage lines, especially in arid and semi-arid environments. Higher mid-slope fuel loads are not visually evident, although they were statistically significant. Also, known fuel load variability was under-represented by classifying fuel load estimates into broader categories.

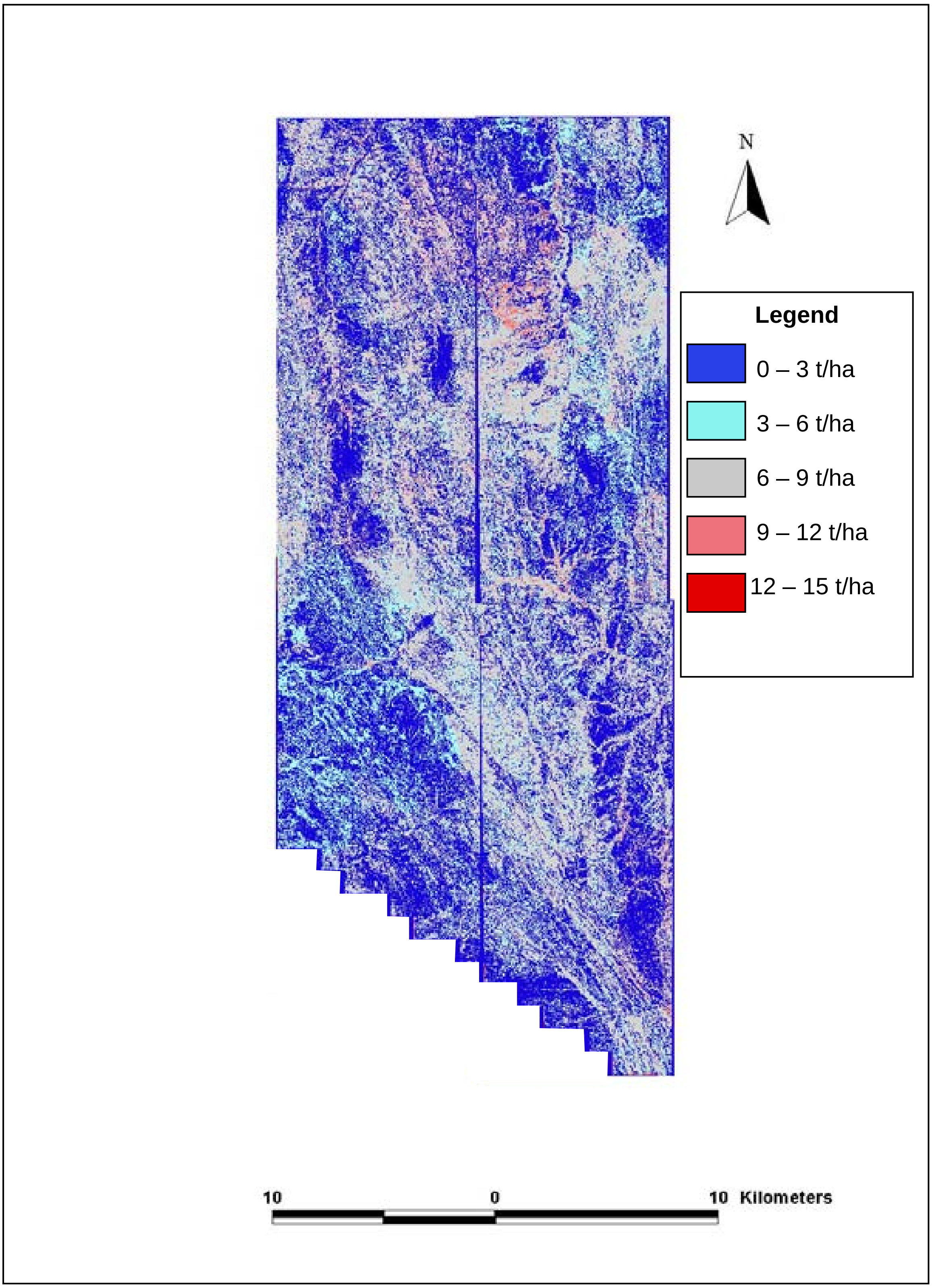

Figure 4.16 Fuel load map for the Calton Hills study area.

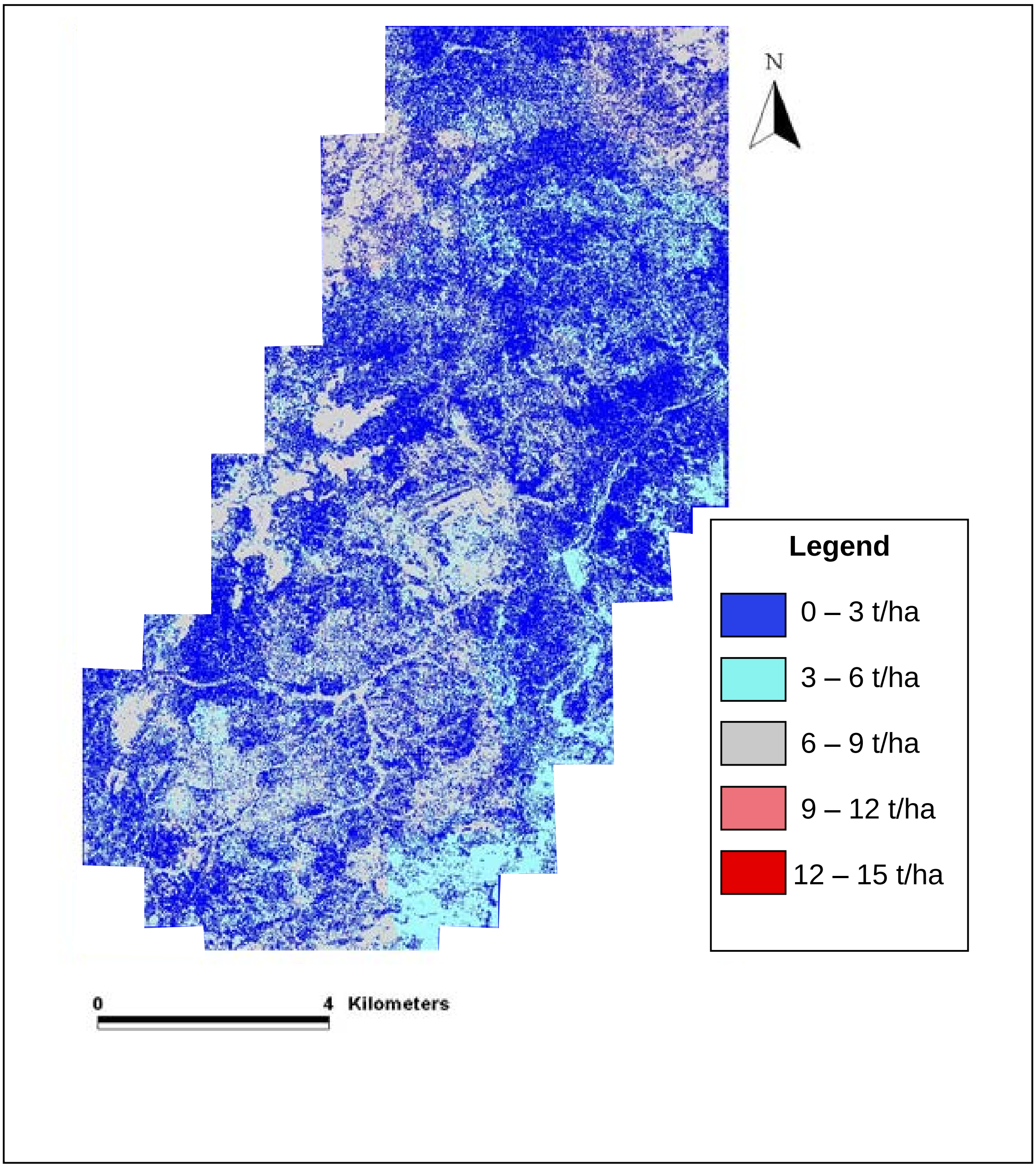

Figure 4.17 Fuel load map for the 'Gunpowder' section of the Calton Hills study area.

Comparing the predicted fuel loads at each fire age with corresponding field estimates identified model bias at low and high fuel loads as expected (Figure 4.18).

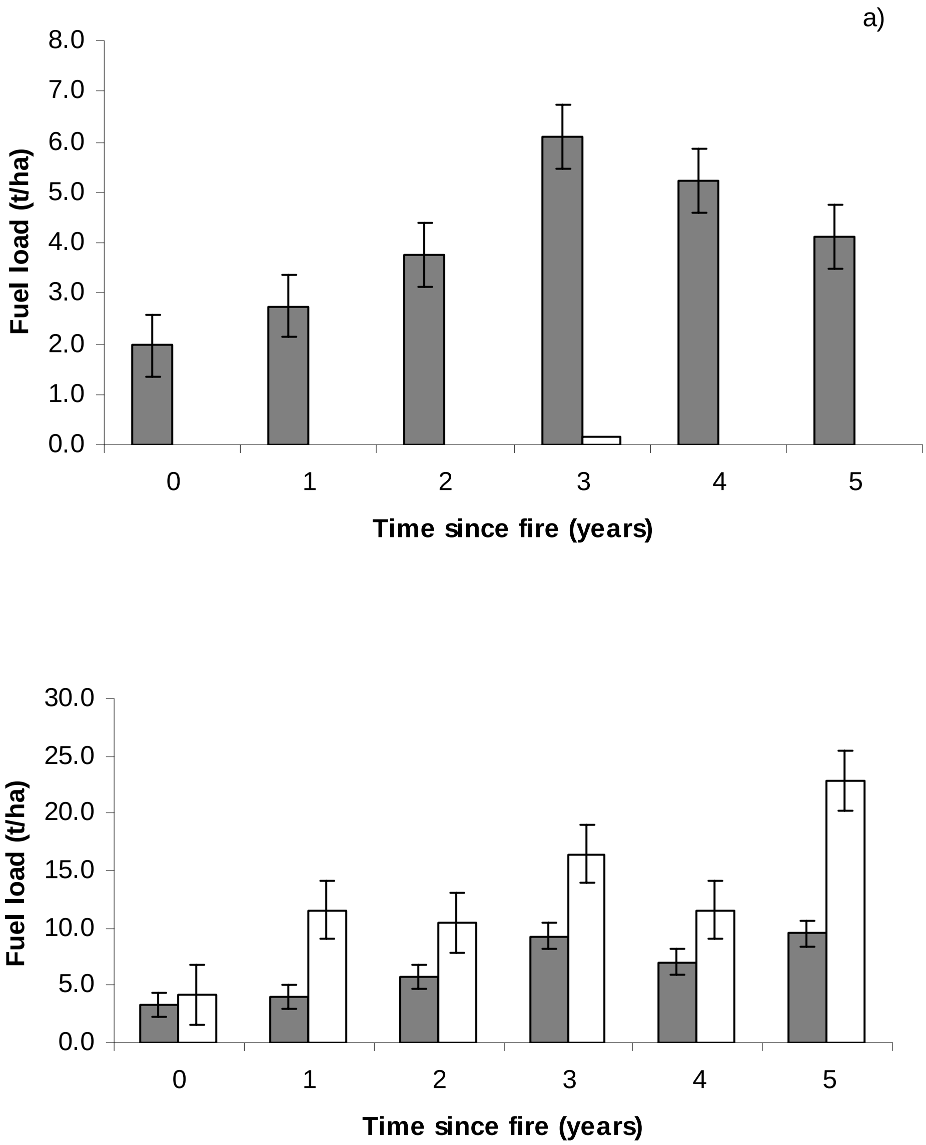

Figure 4.18 a) The regression model over-predicted the minimum fuel load at each fire age. (Grey bars = predicted minimum, white bars = actual minimum, error bars = standard error). b) The model under-predicted the maximum fuel load at each fire age. (Grey bars = predicted maximum, white bars = actual maximum, error bars = standard error).

On average, fuel load was over-predicted by 0.7 to 1.5 t/ha across all fire ages (Figure 4.19). Based on this finding the model was considered reasonable for generating a map to categorise areas of low, medium or high fuel load. The main concern was that areas of very high fuel load, and thus fire hazard, could be overlooked.

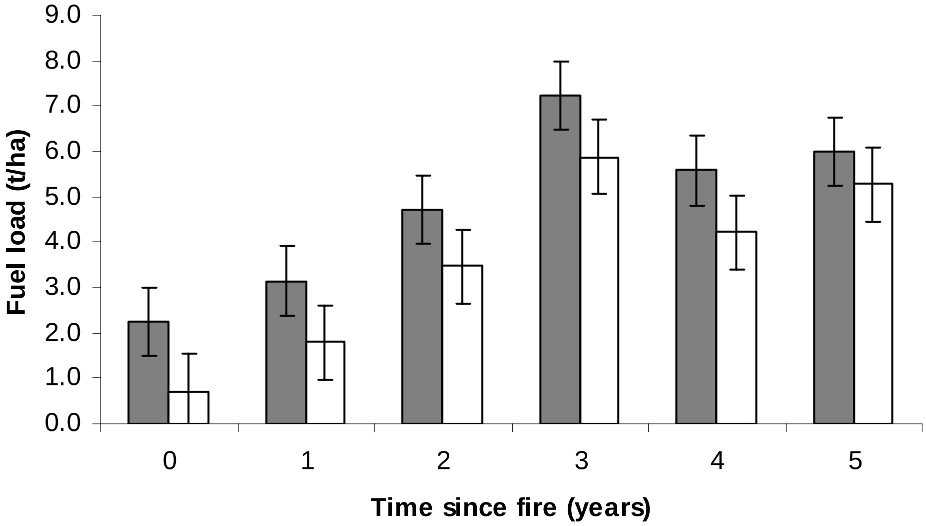

Figure 4.19 On average, the regression model over-predicted fuel load by 0.7 to 1.5 t/ha across all fire ages. (Grey bars = predicted average, white bars = actual average, error bars = standard error).

Model bias can be explained by comparing NDVI and fuel load at each point (Figure 4.20). Points with high fuel load and low NDVI had very dense spinifex which was grey and moribund. Some had senescent *Acacia* as well. In these cases, fuel load would be substantially under-predicted. Points with high NDVI and low fuel load were invariably on bare ground, such as creek banks or river beds. These places, and other areas where fuel was very sparse or absent (<1 t/ha), could still have a moderate or high NDVI, with soil dominating the response. Low fuel and high NDVI may also result where bare ground occurs under a tree canopy. Further, high fuel load estimates were constrained by the limited range of NDVI values. For example, 9.5 t/ha was the maximum value that could be predicted based on the NDVI at each point. The maximum field measure of fuel load was 22.8 t/ha.

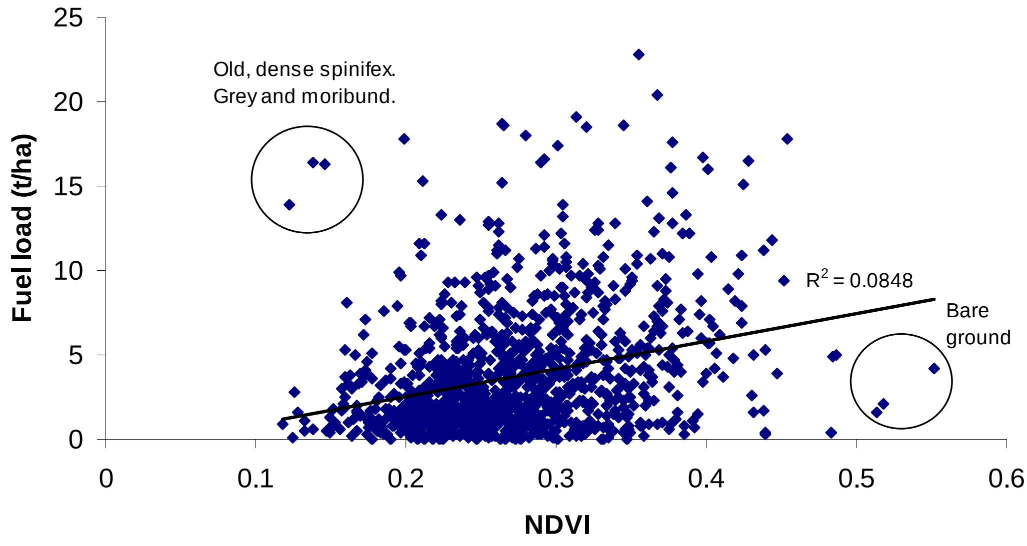

Figure 4.20 Comparing NDVI and fuel load at each point helped to explain model bias: points with old, grey spinifex had high fuel load but low reflectance; bare ground had high reflectance but very low fuel.

When actual and predicted values were compared on a point by point basis, the overall correlation was only 20% (R^2 = 0.2023, $p < 0.05$, n = 794) (Figure 4.21). Scaling issues offered additional explanation. There was considerable fuel load variability at each sampling point. Disc-drop height could range from 0 cm to over 100 cm within the 9-pixel sampling area. The NDVI did not always relate strongly to fuel load because a few very large spinifex hummocks surrounded by bare ground would give a high fuel load and a moderate NDVI. A general, statistically significant relationship was detected, but this was not accurate when compared to the scale of observation, given the discontinuous nature of hummock grasslands. All fire ages had a wide spread of disc-drop heights and thus fuel load estimates (Figure 4.22) and bare ground was plentiful (Figure 4.23). The NDVI measured a broader scale. Model construction was based on averaging and probability, which prevented fuel load mapping at the level of detail anticipated using fine scale imagery. The potential for subsequently using the fuel load map to measure fine scale variability is therefore limited.

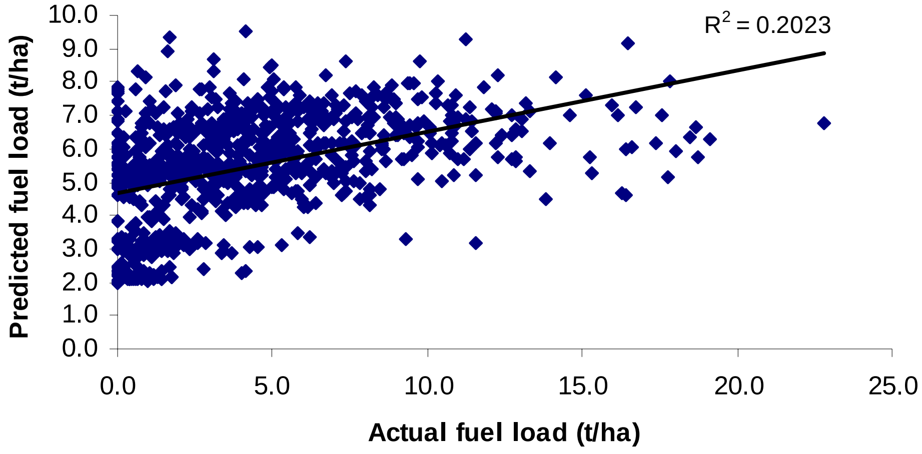

Figure 4.21 The correlation between actual and predicted fuel loads for each point was around 20% (R^2 = 0.2023, $p < 0.05$, n = 794).

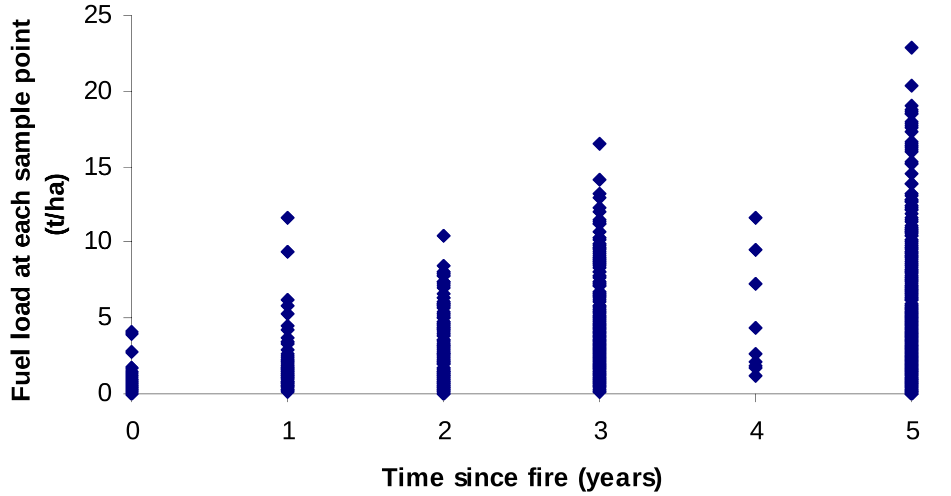

Figure 4.22 Fuel load at each sampling point was based on the average of 9 disc-drop measures. Fuel load varied considerably at each fire age (n = 1,199 disc drops).

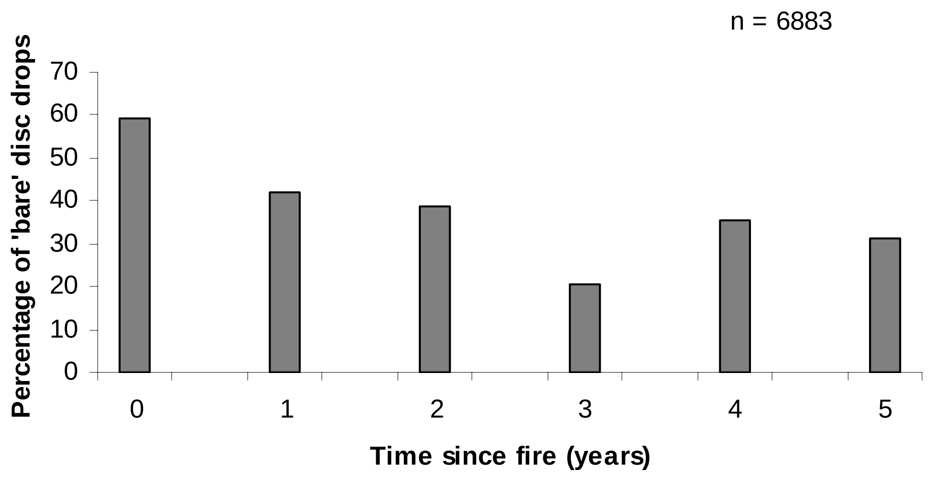

Figure 4.23 Bare ground was present at all fire ages. (n = number of disc drops).

4.3.2 The Post-fire Landscape

Fire severity indicators and reflectance values

Tree species had different mean reflectance values for each band and index, with the most marked differences detected using PVI ($F = 46.92$, $p < 0.001$, $n = 500$). However, this index was not useful for discriminating between shrub species. Shrub species had significantly different mean reflectance values for all bands and indices ($p < 0.001$) except PVI. The strongest relationship for differentiating between shrub species was with EVI, the enhanced vegetation index ($F = 6.066$, $p < 0.001$, $n = 150$).

Tree char height was considered the best indicator of severity (see Chapter 3), but this variable was not strongly related to reflectance. The best relationship for char height was its very weak correlation with PVI (Pearson's $R = 0.093$, $p < 0.04$, $n = 492$).

Twig diameter (Log_{10}) was correlated to all bands and indices (all $p < 0.001$, except for PVI where $p < 0.02$). The strongest relationship for this variable was with band 2 (green) (Pearson's $R = 0.319$, $p < 0.001$, $n = 150$), a band also weakly related to the percentage of tree canopy scorch (Spearman's Rank, $R = 0.139$, $p < 0.002$, $n = 500$). Relationships between shrub charring and reflectance values were weak too. The strongest was with Log_{10} NDVI (Spearman's Rank, $R = 0.216$, $p < 0.007$, $n = 150$).

Point-based severity scores (i.e. the derived index) were not significantly correlated with any band or index. However, when severity scores were used as categorical data, the mean reflectance for all bands and indices differed significantly from one another, although the pattern was not strong (F ranged from 2.67 - 3.8, $p < 0.001$ to 0.007, $n = 500$). The best result was for the green band ($F = 3.8$, $p < 0.001$, $n = 500$).

Fire severity mapping could not be undertaken on the basis of this data. Significant differences in reflectance were detected for different tree and shrub species, and tree and shrub species influenced fire severity (Chapter 3), but the relationships were not sufficiently strong for mapping purposes. Collinearity between variables prevented analysis using multiple regressions.

NDVI differencing and fire severity indicators

There was no correlation between NDVI difference values and tree char height or point severity scores. Therefore fire severity mapping using this method was not carried out.

Inferring fire intensity from fuel load

Intensity values predicted from the combining the fuel load map and digital elevation model using MacArthur's equation were not related to on-ground fire severity measures. Given the very general nature of fuel load predictions based on the multivariate equation (with NDVI as a dependent variable), and lack of information on weather conditions at the time of the fire, the fire severity map using this method was not considered further.

4.4 Discussion

4.4.1 Fuel Load Map

The key finding from this study is that IKONOS satellite imagery can be used to map fuel load in spinifex-dominated woodlands, but using fine scale imagery does not necessarily give a fine scale result. Areas of potentially high wildfire risk can be identified, but on-ground verification of these fuel load maps is needed before making management decisions, or using them for pattern analysis to quantify landscape patchiness (see Gustafson 1998; Fortin & Dale 2005), because their accuracy is limited by a number of factors, including method.

Each major step in the method incurred error. It was associated with image pre-processing, modelling fuel load relationships, using probabilities to define vegetation types, and adopting methods developed in the northern hemisphere for use in semi-arid Australia. Each of these errors is explained further below, but given that error accumulates in a multi-step process, analysis using remotely sensed data will inevitably have uncertain results.

There are many issues yet to be resolved in processing IKONOS imagery. Pixel resolution of 1 to 4-m is promising, but realising the potential depends on the accuracy of other data layers. Law and Nicol (2004) emphasised the significance of using a digital elevation model at a similar scale to the imagery, given its importance for calculating slope, aspect and other variables associated with pre-processing. A 5 m contour map was used in their 25 km^2 Hong Kong study area, but such detailed elevation mapping is not widely available. The digital data available for the Mount Isa study area were limited and their accuracy questionable. For instance, the

1:100,000 road data was based on topographic maps produced in 1971. Some roads no longer existed, or had been realigned. Instead, river junctions were used for registering the images, but these may have changed through erosion. This highlights the importance of ground control points for ensuring that the same spatial relationship exists between datasets (Reinke & Jones 2006). I recorded a number of ground control points using a hand-held GPS, but this in itself had +/- 5 m accuracy. Registration offsets of a few metres can affect the relationship between field plots and remotely sensed data (Reinke & Jones 2006).

There is still debate over the best way to correct images for topographic shadowing. A number of methods are available, but these are still being tested in different environments (Law & Nichol 2005; McDonald *et al.* 2000; Riano *et al.* 2003; Wu *et al.* 2004). Their applicability to IKONOS imagery has not been established, although Law and Nicol (2004) found that 'normalisation' worked better than 'cosine correction' to improve a small area of mountainous terrain. The cosine correction worked better in the Mount Isa area for a sub-set of the imagery. Its use was more straightforward, but correction techniques need further testing.

Ancillary digital data are critical for making the best use of IKONOS imagery. This was a key limitation in the Mount Isa area. The only published digital vegetation map was at 1:1,000,000 scale (Fox *et al.* 2001) and no soil maps were available. Soil mapping at an appropriate scale may have allowed for improved site stratification. Given the effect of soil on the NDVI response in Australian rangelands (Graetz & Gentle 1982), accounting for soil surface reflectance effects would likely improve fuel load differentiation.

My findings underscore the recognised limitations of the NDVI in arid and semi-arid Australia. Bare soil and senescent vegetation influence the NDVI 'greenness' signature (Link *et al.* 2006), which constrains its use for mapping spinifex fuel load. The enhanced vegetation index (EVI) was proposed to minimise the soil effect (Harris *et al.* 2006) but, when tested here, EVI had a weaker relationship with measured spinifex fuel load than NDVI. Thus, site ‘averaging’ was accepted in order to compile the fuel load map, but the result was not at the fine scale anticipated for by using IKONOS imagery.

The validity of any model depends on data inputs. In this case, the fuel load at each point was itself based on a modelled relationship (fuel load was predicted from disc-drop height), which had associated error. Similarly, fire age was inexact. Each age covered a 12-month time span, as well as effects of patchiness within fire scars. Some sites could have had up to one year of extra growth, so fuel load variation within and between different fire ages was expected. A stronger relationship may have been detected using non-linear modelling alternatives, but further investigation is required. Finally, the model and resultant map can only be considered probabilistic, given the ‘cut-off’ values assigned to NDVI for spinifex and to the green band for *Acacia*. These mapping decisions meant some areas with spinifex were not included on the map (where NDVI was < 0.264), some areas mapped as spinifex were ‘other grass’, and fuel load may have been assigned to *Acacia* where it was absent and vice versa. These points emphasise that applying fine scale imagery to ecological problems may be conceptually promising, but image analysis techniques are yet to be refined and the reality of being able to effectively capture on-ground variation is problematic.

Despite these limitations, the fuel map produced using IKONOS imagery is an advance in two ways. First, it is at a finer scale than produced previously. Any fuel map prepared from satellite imagery, at every scale, contends with similar shortcomings; each scale will depict general patterns at the level being considered. Second, differences in fuel load were detected within the same ground cover type. Where remotely sensed data have been used elsewhere to map fuels, the principal aim has been to make comparisons between regions and sub-regions (Burgan *et al.* 1998; Dymond *et al.* 2004; van Wagtendonk & Root 2003). There is criticism that these effectively only mimic vegetation maps (Keane *et al.* 2001). This comment also applies to the single publication where a fuel map was produced using IKONOS imagery (http://www.spaceimaging.com/products/wfras). The image was used like an aerial photograph to identify vegetation types, and areas of high fuel load were determined though expert opinion.

A major requirement is to specify the intended use for a fuel load map. Broad scale studies to identify wildfire hazard areas for regional planning require a different approach than finer scaled studies aimed at detecting localised variability, which may be important to species conservation. Essentially, the question should dictate the scale of analysis required (Stone & Haywood 2006; Zerger *et al.* 2006). IKONOS imagery promises a certain level of detail, but has a number of drawbacks. It is expensive and not necessarily available for specific time periods. It also requires significant computer processing power and expertise to move from the format provided to a useable end product with the requisite information. Therefore, its use for monitoring landscape patchiness as a standard practice is very limited, but could be justified in special circumstances, such as fire management for endangered species conservation. For this it is important to understand the assumptions that underpin the maps and recognise the inherent limitations of remotely sensed data. Determining the necessary level of detail in mapping is a critical first step.

Fire severity mapping

Attempts to map fire severity were less successful than fuel load mapping. In addition to the issues described above, the time lapsed between fire occurrence and image capture hindered analysis. Hammill and Bradstock (2006) successfully mapped fire severity from Landsat data recorded approximately 2 months post fire, with field data recorded 12 to 26 months post fire. The image used here was captured nearly 5 months post fire, with field indicators of fire severity recorded six months post fire. Although fire severity indicators, such as char height on tree trunks, were clearly evident on the ground, there had been an intervening wet season with an associated flush of vegetative growth. Vigorous post-fire growth greatly affects surface reflectance, hence fire severity and reflectance values were poorly correlated. This highlights an important issue. In most circumstances, analysis based on remotely sensed data is improved where the dates of image capture and field measurements are closely matched. But for fire severity mapping it is very important to have remotely sensed data recorded as close as possible to the burn date. Fire effects can be discerned for some time, so later field measurements are acceptable.

NDVI differencing is one of the main methods used to map fire severity (Hammill & Bradstock 2006). However, this procedure is inaccurate where fires occur after vegetation has died and in areas with sparse vegetation before the fire (Cocke *et al.* 2005). Therefore, NDVI differencing may be inappropriate for use in the Mt. Isa semi-arid environment because of the grey colour of old spinifex, because the *Acacia* shrubs are relatively short-lived, and because bare ground is plentiful. However, this procedure should be tested further using a planned fire, before and after field measurements, and well matched dates for fire occurrence, image capture and field survey. In other areas the ‘normalised burn ratio’ has been adopted to overcome issues associated with NDVI (Cocke *et al.* 2005), but the ratio is based on Landsat Band 4 and Band 7 (near- and mid-infrared, respectively). Mid-infrared data are not captured by the IKONOS sensors.

The modelling approach for predicting fire severity requires more work. Again, such investigation should be based on a planned fire. Detailed weather data for the duration of the fire is indispensable for sound interpretation.

Implications for research and management

From a research perspective, there are ample opportunities to improve how remotely sensed data, especially IKONOS imagery, can be effectively applied to ecological fire management. Of greatest priority is the application of remotely sensed data to a wider range of ecological questions. These data provide alternative insights to landscape structure and function. The potential to expand from plot-based studies to entire landscapes is also tantalizing. A critical body of literature will enable the best use to be made of these newly available tools. Even basic level research will help. For example, different image pre-processing methods need to be tested in a range of environments. A universal solution may not exist, but knowing which technique is appropriate for given circumstances would be an advance. This also applies to the range of available vegetation indices. The various approaches should be assessed in the course of applied studies, however. They add to uncertainty but do not prevent imagery use.

Results here demonstrate that IKONOS imagery can be used to map fuel load at a scale that records variation within semi-arid grassy woodland or savanna plots. But there are still problems with generalisation in the face of heterogeneity. Site 'averaging' is inevitable. This emphasises that no single scale of monitoring or application of remote sensing is best. Research approaches that span a range of scales are desirable for addressing problems at a particular 'focal' scale (Ludwig *et al.* 2007).

More work is required to effectively map fire severity. This issue continues to thwart ecological research and may still be possible using IKONOS imagery. Targeted research with well timed image-capture and consideration to data compatibility between scales would be an important first step (Reinke & Jones 2006). Acquisition of imagery for specific dates or months is problematic in northern Australia due to prevalent cloud cover. Use of airborne sensors may partially overcome this problem. However, it must be appreciated that the level of detail sought by researchers may not be necessary to broader levels of fire management.

For managers, this study reinforces the point that concentrating on fire size is a realistic option for ecological fire management in the Mount Isa Inlier. Fuel load varied widely at every fire age. This high level of fuel heterogeneity suggests a degree of resilience: every fire will have unburnt patches. Defining a desired level of internal fire patchiness is problematic. There are also implementation difficulties in spinifex woodlands, and monitoring internal patchiness at a fine scale is not yet practical. It is more sensible to accept or acknowledge intrinsic spatial heterogeneity, define an optimal fire size, and then efficiently monitor results at the landscape scale using established and readily available techniques.

Fire was demonstrated to be the key driver of fuel load in spinifex woodlands, with fuel load highest on long-unburnt areas. High fuel loads are associated with increased wildfire risk, but wildfire risk can be reduced by the proactive use of fire. Areas long-unburnt can be readily identified from coarse-scale fire history maps. Thus, fine-scale fuel load maps are not necessary unless fire management is being targeted to small areas, such as under high-voltage transmission lines. Note that I do not advocate burning all long-unburnt areas. Burning depends on whether protection from wildfire is a management goal for a particular area.

As a general principle, this study showed that areas with spinifex ground cover had higher fuel loads than areas where other grass types were dominant. Data analysis showed that fuel load was higher on mid-slopes than elsewhere, although the resultant map indicated that drainage lines had higher fuel loads. Until this is clarified by further research, both these areas should be considered when planning to implement fire management activities that involve predicting (or containing) fire spread. Drainage lines should be assessed in particular. In wetter climates, drainage lines are customarily associated with reduced fire hazard.

4.5 Conclusion

Remotely sensed satellite data provide additional information for understanding landscape structure, function and dynamics. Satellite technology is becoming increasingly more sophisticated, with data more readily available at increasingly finer resolutions. This has significant appeal for extrapolating ecological interpretations from small field plots to broader landscape scales. However, uncertainty is inherent to remote sensing processes, so results will be generalised regardless of scale. There is considerable research required before fine scale imagery, such as IKONOS, is routinely available and effectively applied. Nonetheless, it can be used to inform fire management, even if indirectly. This study supports the concept of managing fire size in northwest Queensland, rather than aiming for internal fire-patchiness. The most appropriate fire size for species conservation still needs to be determined.

5. MANAGING FIRE TO CONSERVE VERTEBRATE FAUNA

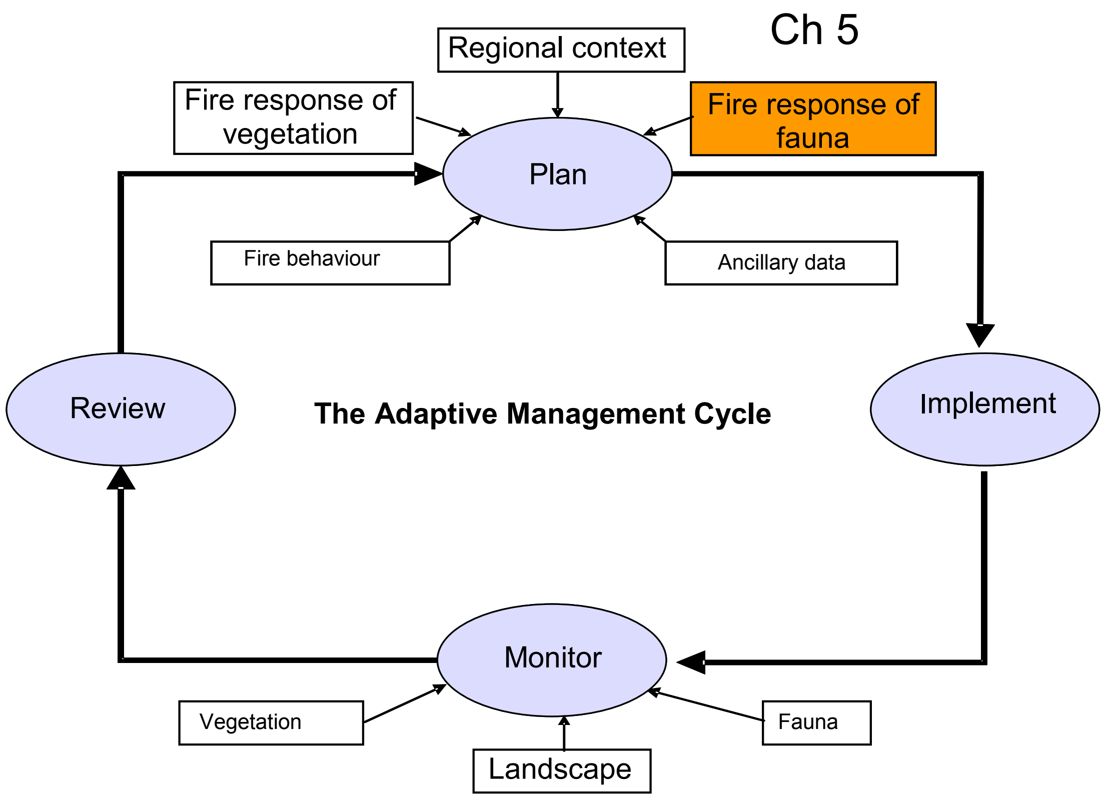

Figure 5.1 Ecological fire management requires consideration of the fauna present and the likely effects of fire. A decision framework to use when species have conflicting requirements would also assist.

5.1 Introduction

Three major questions confront managers who are planning fires to meet fauna conservation objectives. What fauna are present? How will they be affected by fire? And, which species should be given precedence where there are conflicting requirements? Addressing these questions will assist with setting clear management objectives (Figure 5.1). However, there is a dearth of pertinent data available to address these questions. There are many species to consider and fire responses will be different depending on timing and other fire characteristics (Whelan 1995; Whelan *et al.* 2002). Formal studies are exceedingly few and may not be immediately applicable. For example, results from single, small-area studies in temperate or mesic regions may not be transferable to species in a tropical semi-arid environment.

This chapter presents a process for the Mount Isa Inlier that could be adopted for use at a range of management scales and is likely to be applicable to other geographic areas. It addresses the complexities presented by different study sizes, locations, areas and foci. It also addresses the shortage of empirical studies by capturing 'expert opinion'. This is a valid method of assembling scientific information for use in qualitative or quantitative analysis, but certain protocols are required to minimise bias (Burgman 2005). In the current context, it provided an efficient approach to rapidly amass information on potential fauna responses to fire. These data were used to identify fire-sensitive species.

However, a process was still required to guide the setting of management objectives. Three options were considered. First, species were categorised similar to Lambeck's (1999) 'focal species' approach. Consideration was given to whether they would be area limited, resource limited or dispersal limited as a result of fire. Secondly, the functional groups proposed by Bradstock *et al.* (2005) were applied. These are based on species refugia requirements during a fire and the recovery stages they utilize after a fire. Finally, the concept of 'increaser' and 'decreaser' species (Landsberg *et al*. 1997) was applied. The predicted response of species to burning every five years, every twenty years, or to no fire was investigated.

Ideally, the categorization process would reliably and objectively identify the fire-sensitivity of species. The specific requirements of the most sensitive species could then be addressed when planning the use of fire. To test the theoretical predictions, birds were chosen as a faunal sub-group and their predicted responses to fire were tested in the field by conducting surveys in burnt and unburnt areas. Birds were used for ease of census and because they are responsive to land management practices (Landsberg *et al*. 1997; Mac Nally *et al*. 2004). Birds have also been recommended as indicators for environmental monitoring at landscape and bioregional scales (Smyth & James 2004).

Five fire-sensitive bird species were investigated further to identify their fire patchiness requirements. The results are discussed in terms of desirable fire size and fire frequency, with suggestions on how management outcomes might be monitored at the landscape scale using satellite imagery and on-ground surveys of target species. The adequacy of current fire patterns for vertebrate fauna conservation in northwest Queensland is also discussed.

5.2 Methods

5.2.1 Predicted Fauna Responses to Fire

A database was constructed to collate information on the likely response of terrestrial vertebrate species to fire. A number of data fields relating to direct and indirect fire responses were established. These covered potential effects of individual fires, as well as effects of the overall fire regime operating over time. The intention was to secure as much information as possible so different classification options could be investigated. A list of potential responses was compiled for each data field, where applicable, to ensure consistency of response and terminology (Table 5.1).

Table 5.1 The data fields and categories used to collate information on the likely response of vertebrate species to fire, and potential response options.

Habitat	Fire dependency of vegetation	Feeding guild	Home range	Immediate response to fire	Response after fire
Low open-woodland with spinifex understorey	Requires infrequent fire (fire-dependent associations)	Terrestrial granivore	Unknown	Burrowing	Recolonise
Acacia dominated forest, woodland and shrubland	Damaged by too-intense fires (Fire sensitive associations)	Arboreal granivore	Nomadic	Shelter (logs, rocks, cracks)	Move between burnt/unburnt
Eucalypt open-forests - woodlands	Tolerates regular burning (Fire-tolerant associations)	Frugivore	<1ha	Shelter (trees)	Persist only in unburnt habitat
Riparian/floodplain eucalypt open-forests - woodlands	Damaged by frequent, too intense fires (Fire-sensitive associations)	Herbivore	1-5 ha	Limited ability to disperse	Needs given size unburnt
Vine thicket	Damaged by all fires (Fire-sensitive associations)	Nectarivore	5-10 ha	Moderate ability to disperse	Tolerate a range of conditions
Wetlands (Riparian areas, swamps, lakes)	Fire-tolerant associations (or n/a)	Frugi., nectari., insecti.	10-100 ha	Can disperse broadly	Feeds on early recovery stages
Multiple habitats	Tolerates infrequent fires	Nectari., insecti.	>100 ha	NA (aquatic)	Unknown
Rock outcrops		Frugi., insecti.			
Tussock grasses (not Mitchell)		Carnivore			
Tussock grassland (Mitchell grass)		Aerial insectivore			
		Terrestrial insectivore			
		Arboreal insectivore			
		Omnivore			

Table 5.1 (continued)

Likely fire effects	General fire tolerance	Patchiness requirement	Fire cycle sensitivity	Particular vulnerabilities	Seasonal intolerance	Non-fire related factors influencing abundance	Study area location
Killed by fire	Tolerant	n/a	Nomadic opportunist attracted by fire	Requires long-unburnt areas	Early wet season	Predation by feral animals	Mesic savannas
Nil	Intolerant	Depend on known burnt/unburnt combination	Adapted to forage in post-fire environment	Vulnerable at nesting time	Late wet season	Habitat loss (clearing)	Semi-arid, tropical savannas
Post-fire predation	Fire dependent	Depend on unknown burnt/unburnt combination	Adapted to short fire cycle	Hollows vulnerable	Early dry	Habitat loss (weed invasion)	Semi-arid, temperate savannas
Decline from loss of groundcover and shelter	Partial protection of habitat required	High degree unburnt (>40%)	Adapted to moderate fire cycle	Loss of ground cover	Late dry	Strength of wet season	Temperate
Long term structural and floristic modification		Moderate degree unburnt(10-40%)	Adapted to long fire cycle	Loss of shrub layer	Unknown	Intensity of grazing	Arid
Generally benefits from fire		Tolerates low level unburnt (<10%)	Not sensitive	Loss of tree layer		Unknown	
Unknown		Unknown	Unknown	Loss of mistletoe		Other	
				None			
				Unknown			

Table 5.1 (continued)

Study area size	General fire group	** Bradstock *et al.* classification	Focal species classification	Gut feeling- landscape fire age	Burnt 5 years	Burnt 20 yrs	Unburnt
<1ha	Requires frequent fire	RpRtSpSt	Area limited - need minimum unburnt area to sustain population	<1yr	1 = become extinct from the landscape	1 = become extinct from the landscape	1 = become extinct from the landscape
1-5 ha	Tolerates frequent fire, if low intensity	RpRtSp	Dispersal limited - require minimum distance to unburnt resources	1-5 yrs	2 = persist	2 = persist	2 = persist
5-10 ha	Tolerates occasional intense wildfire	RpRtSt	Resource limited - fire likely to affect food/nesting resources	6-10 yrs	3 = optimal	3 = optimal	3 = optimal
10-100 ha	Tolerates low intensity, patchy fires	RpSpSt	Limitations other than fire	10-20 yrs			
>100 ha	Requires long fire free interval	RtSpSt		>20 yrs			
	Tolerates a wide range of fire intensity and fire frequency	RpSp		Not relevant	** Rp: refugia permanently available during fire (e.g. rocky areas);		
	Unknown	RpSt	Note: Limitations other than fire also includes 'unknown'	Never	Rt: refugia transient (e.g. unburnt patches) during fire;		
	Requires burnt/unburnt in combination	RtSp			Sp: inter-fire habitat suitable at all stages;		
		RtSt			St: inter-fire habitat only suitable at certain stages		

These lists were presented in menu format and linked to relevant spreadsheet cells to facilitate data entry. The following rationale was used for each data field:

- **Habitat**: These are the broad vegetation types in Queensland's northwest highlands. The option 'multiple' was used if the species was not associated with a particular habitat.
- **Fire dependency of vegetation**: This addressed secondary fire effects, where habitat change over time can affect the occurrence of species (e.g. vegetation thickening in the absence of fire contributing to the decline of grassland birds (Crowley & Garnett 1998)).
- **Feeding guild**: Fires can affect the plant species present and the abundance of their flowers, fruits and seeds, and the abundance of prey populations and their ease of capture (Franklin *et al*. 2000; Woinarski & Recher 1997).
- **Home range**: The area normally occupied by an individual of a species was used to gauge size requirements for burnt or unburnt areas, and to provide an indication of a species potential for dispersal. It was acknowledged that formal studies are scant and that home ranges can vary depending on resource 'richness' (Woinarski *et al.* 2005). An 'educated opinion' would at least provide a starting point.
- **Immediate response to fire**: What does the animal do in the face of fire?
- **Response after fire**: What does the animal do in the months immediately post fire?
- **Likely fire effects**: This was to capture a general impression on how the species would fare. 'Nil' differed from 'unknown'. 'Nil' meant there would be no effect, such as for a frog living in a waterhole, whereas 'unknown' could apply

where a species initially survives but could be indirectly affected by an unknown factor (e.g. a raptor relying on other birds or small mammals as prey).

- **General fire tolerance**; **Seasonal intolerance; Particular vulnerabilities**; **and Non-fire related factors influencing abundance**: The options in these categories are self explanatory.
- **Fire cycle sensitivity**: This category related to the fire interval. There was a distinction between species that actively seek fire, such as raptors, and species that are adapted to forage for prey in a post-fire environment and granivorous birds that feed on seeds exposed on recently burnt ground (Woinarski & Recher 1997).
- **Reference; Study area location; Information reliability**: These fields were included to ascertain the scope of available research. The aim was to enable data verification and gauge the application of research findings to different areas.
- **Comments; General fire group; Patchiness requirements; and general impression**: These were general categories to elicit information where previous categories did not fit directly. They allowed contributing experts to provide an opinion despite a large number of 'unknowns'.

Two iterations were used to establish the database structure. The first attempt was based on a register of fauna-fire issues in published articles. Associated lists were compiled to summarise response options into a manageable number of variables. Then a pilot exercise was conducted with two experts (John Woinarski and Gabriel Crowley) to review the database structure and response options, and to identify potential shortcomings. This resulted in minor amendments, such as including a 'general impression' category.

A species list was obtained for Queensland's Northwest Highlands bioregion (WildNet, Queensland Environmental Protection Agency). This is comparable to the

Mount Isa Inlier using the national classification system (Thackway & Cresswell 1995). The list gave the species names, common names and conservation status for each class of vertebrate recorded in the area. As very few fauna surveys have been undertaken in northwest Queensland, some entries were based on single sightings. The list provided a baseline, however, and was used to commence filling the database.

I compiled a register of people with a professional interest in fauna and fire and who had worked in the tropical savannas, had publications in peer-reviewed journals and who were considered specialists in their field of expertise. I made personal contact to explain the project and to invite their participation. Eight experts agreed to assist and between them, all four terrestrial vertebrate classes were covered (Table 5.2).

Table 5.2 Names and affiliation of people who contributed their expertise to the fauna fire-response database.

Name	Affiliation	Expertise provided
Dr. John Woinarski	Department of Planning, Infrastructure and Environment, Northern Territory	Birds, general information
Dr. Graham Harrington	Sustainable Ecosystems, CSIRO	Birds
Dr. Mike Mathieson	Environmental Protection Agency, Queensland	Birds
Dr. Geoff Smith	Environmental Protection Agency, Queensland	Birds
Dr. John Winter	Environmental Protection Agency, Queensland	Mammals (other than bats)
Dr. Chris Clague	Environmental Protection Agency, Queensland	Bats
Keith MacDonald	Environmental Protection Agency, Queensland	Reptiles and amphibians
Dr. Gabriel Crowley	Tropical Savannas CRC	Birds, general information

A document was created to outline the intended use of the database and explain the menu options available in each category. It gave instruction on the desired level of response and clarified potential uncertainty due to wording. For example, it was important to remind respondents to consider species at the population level, not as individual animals. Instructions were also given on how to address ambiguity or add additional choices to the menu if certain options were consistently required but unavailable. The database shell and instructions were provided to the experts; half contributed their input through e-mail correspondence and telephone conversations, the others were consulted face-to-face. For the latter, I was able to verbally clarify the intent of each category, provide examples, and reach consensus on the most appropriate option. Instances where experts gave conflicting advice were identified. These were discussed further with the relevant parties until agreement was reached.

The data were used to test the application of different classification approaches for informing fire management. Species were classified similar to i) Bradstock *et al.* (2005) and ii) Lambeck (1999). The 'Bradstock classification' provided 9 options. **Rp** referred to refugia being permanently available during fire (e.g. rocky areas); **Rt** referred to transient refugia during fire (e.g. unburnt patches); **Sp** was where inter-fire habitat was suitable at all stages; and **St** was when inter-fire habitat was only suitable at certain stages. Different combinations of these variables indicated the most likely response to fire. The least fire sensitive species were classified **RpRtSpSt,** while the most fire sensitive species were **RtSt.** These species relied on temporary refugia during the fire and the availability of a particular stage of vegetation recovery post fire.

The 'Lambeck classification' was designed for fragmented landscapes and provided three options. Species could be **area limited** if a minimum unburnt area was required to sustain a population; **dispersal limited** if they required a minimum distance between unburnt areas; or **resource limited** if fire was likely to affect food and nesting resources. A fourth option was added where **limitations other than fire** were the key determinants of population survival.

Finally, the concept of increaser and decreaser species was investigated for birds (Landsberg *et al*. 1997). Three fire frequency scenarios were posed (burning every five years, twenty years, or never) and species were scored according to whether their populations would (1) decline, leading to eventual extinction, (2) persist sub-optimally in the landscape, or (3) increase to an optimal level. The hypothetical scenarios did not consider different fire conditions and timing, but provided a simple construct to assist with evaluating options.

5.2.2 Field Evaluation of Predicted Fire Response: Birds

Field surveys were conducted to compare bird assemblages at burnt and unburnt sites. Six fire scars were identified from satellite-derived data (http://www.firenorth.org.au). Bird composition and abundance was surveyed in June 2005 on sites located within each fire scar. Site selection was based on fire size, vegetation type and access. Emphasis was on large fires in order to minimise edge effects. Spinifex/snappy gum communities were selected to complement earlier field work. Each area had burnt the previous November except one, which had burnt in February 2005 (Figure 5.2).

Survey methods were based on Watson (2003), except that surveys were timed (1 hour each). The 1 hour time allocation was based on the personal experience of one of the surveyors (Graham Harrington). Watson (2003) recommended a standardized search effort, stopping when additional records did not add significantly to diversity estimates. This ‘stopping rule’ was precluded by logistical constraints, however; the 1 hour surveys were adopted after a pilot study established that this time allowance provided a robust method for comparing sites.

Two pairs of field workers undertook the bird surveys. Each pair consisted of a trained observer and a scribe. Surveys were conducted in the morning and afternoon at each fire scar: one pair commenced 500 m within the burnt area; the other pair commenced 500 m away from the fire scar edge. After the 1 hour survey effort, each pair moved to the alternative burnt/unburnt area and undertook a second survey. Thus, every burnt and unburnt area was sampled four times, twice in the morning and twice in the afternoon.

Surveys involved navigating to a site, noting time and location details as birds settled, and then recording the birds seen or heard. Records were made while walking into or away from a fire scar, depending on whether it was a burnt or unburnt site. Walking direction was haphazard, however, as unfamiliar calls were actively pursued. All birds seen or heard were noted, along with time of detection and supplementary data such as proximity to drainage lines or unburnt spinifex patches. Birds flying over were noted but were not included in data analyses.

Two of the species predicted as fire sensitive were also known to be cryptic and difficult to detect. Therefore Carpentarian Grasswrens (*Amytornis dorotheae*) and Kalkadoon Grasswrens (*Amytornis ballarae*) were actively sought using the 'play-back' technique (Gibbons & Gregory 1996). This method was incorporated into the standard survey and involved stopping every 10 minutes to play recorded calls of these species. Calls were played using a Sony® mini-disc player (model MZ-NH600) with external Creative® Travel Sound speakers. Calls were projected at a fixed volume while making a 360^0 sweep around the stopping point. Birds responding to the calls were recorded.

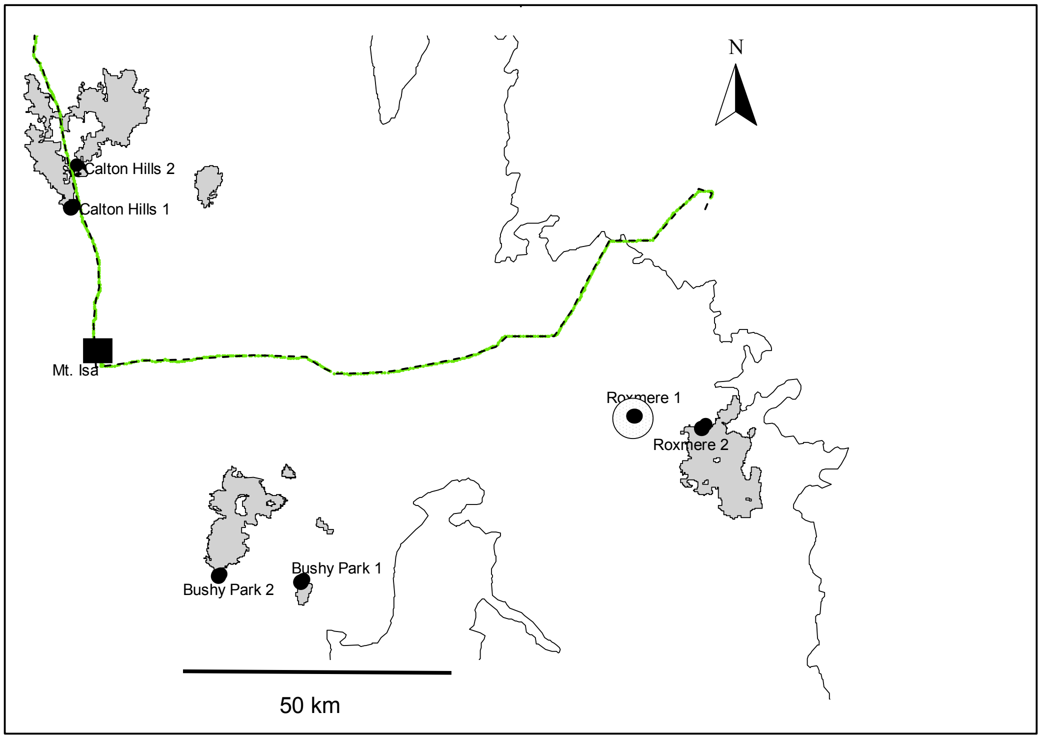

Figure 5.2 Map showing the six sites (black dots) where bird surveys were undertaken inside and outside fire scars (grey). (Dashed line is powerline access track; Fire scar boundary was not available for Roxmere 1).

Data were analysed as follows:

- Paired t-tests were used to compare the total number of bird species, and bird abundance, between burnt and unburnt sites. Comparisons were also made between morning and afternoon surveys.
- A series of site ordinations (Non-metric Multi-dimensional Scaling, McCune & Grace 2002) were undertaken based on bird presence/absence and abundance data to investigate species compositional differences between burnt and unburnt sites. Initially differences were investigated on a sample by sample basis since combining samples can mask subtle differences (Mills 2004). Thereafter, visits were grouped at the fire scar and property levels (i.e., sites within the six fire scars were evenly spread between 3 properties).

- Indicator species analysis was carried out based on the method of Dufrene & Legendre (1997) using the software PC-ORD v. 4.0 (McCune & Grace 2002). This method calculates the relative abundance and relative frequency of each species and combines these values to compute an 'indicator value' for the variables being considered. Levels of significance are calculated using a Monte Carlo procedure. The indicator species suggested by this technique were compared with species predicted as being fire sensitive in the fire response database.
- A visual inspection of the data suggested that five species were strongly associated with unburnt spinifex. These were: Variegated Fairywrens (*Malurus lamberti*), Kalkadoon Grasswrens (*Amytornis ballarae*), Spinifexbirds (*Eremiornis carteri*), Magpies (*Gymnorhina tibicen*) and Mistletoebirds (*Dicaeum hirundinaceum*). Chi-square tests were used to determine if their habitat preference was statistically significant.

5.2.3 Determining Fire Patchiness Requirements for Target Species

Field evaluation of predicted fire responses confirmed the fire-sensitivity of some species. This supported the notion that, if internal fire patchiness requirements could be determined for these species, it would assist with setting quantitative fire management objectives and associated monitoring. Five target species, all sedentary, territorial, ground-dwelling birds, were identified for further study. These were Variegated Fairywrens, White-winged Fairywrens, Kalkadoon Grasswrens, Carpentarian Grasswrens and Spinifexbirds.

Bird calls were obtained for these species (Bird Observers Club of Australia) and recorded onto mini-discs. A field survey was undertaken in July 2005 focusing on areas that had burnt in the previous 12 months (Figure 5.3). Three observers recorded data in the following way. First, a fire scar was located and visually assessed. Then, nearby unburnt areas (> 7 years since fire) matching in terrain and vegetation were selected. Sampling commenced in the unburnt area after a paired burnt area was chosen. Observers were well separated from one another and walked over a broad area, stopping to play the bird calls at 10 minute intervals. Sampling continued for one hour. If any of the target species was confirmed in the unburnt area, the burnt area was then surveyed. Again, the three observers walked in different directions, this time seeking unburnt patches within the fire scar. Unburnt patches were sighted from a distance and approached quietly before playing the mini-disc. If a target species appeared, records were made on the number of individuals, time of sighting, exact location (using a hand held GPS) and patch characteristics. Six patch characteristics were recorded: (i) approximate size (m^2); (ii) estimated distance to the next unburnt patch (m); (iii) estimated distance to the fire scar edge (m); (iv) number of unburnt patches in a 50 m radius; (v) spinifex clump density (on a 1 - 4 scale from isolated to close packed); and (vi) spinifex type and height (m). In total, 15 - 18 unburnt patches were sampled in each of the 11 recently burnt areas. The total survey time was 66 hours, 33 hours in unburnt country and 33 hours within fire scars.

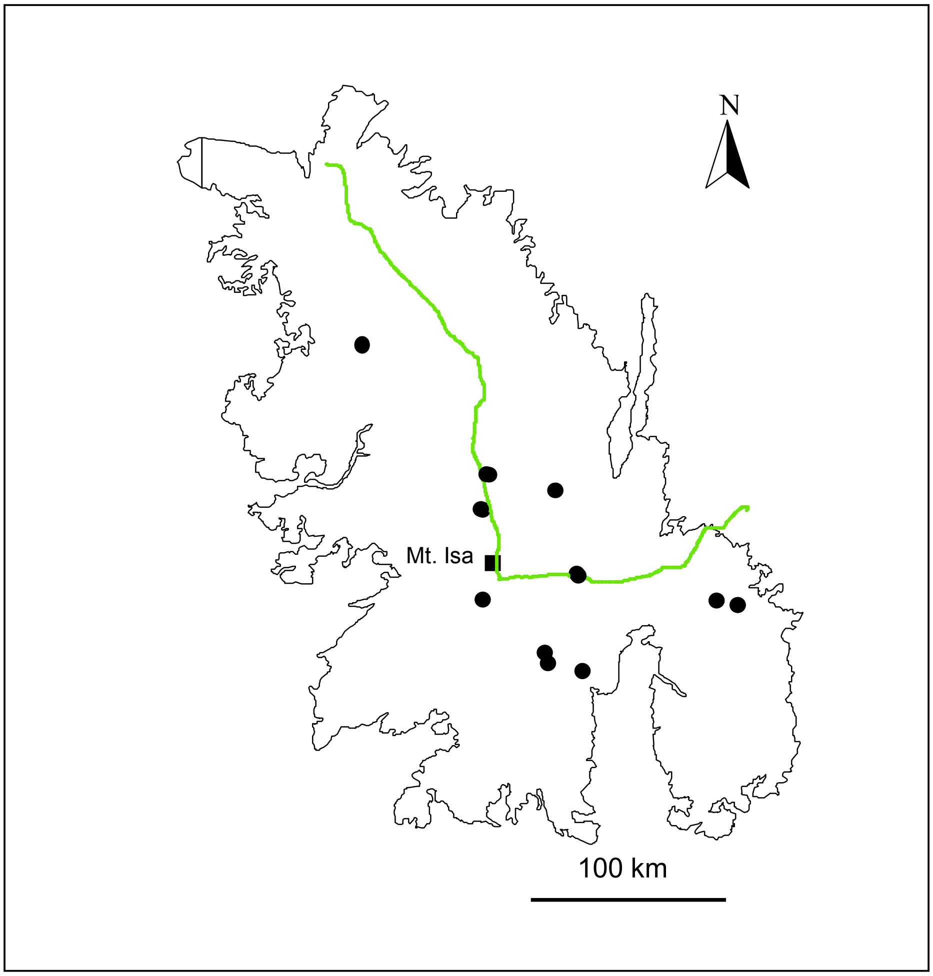

Figure 5.3 **Eleven sites (dots) within the Mt. Isa Inlier where surveys were undertaken to investigate the internal fire-patchiness requirements of fire sensitive bird species. All sites had burnt within the previous 12 months. (Green line represents the powerline access track).**

5.3 Results

5.3.1 Fauna Database Analyses

The completed fauna database is presented in the enclosed CD, Appendix 4. There was strong consistency in expert opinion for most species. Inconsistencies arose when the fire response of individual animals was considered, rather than the overall population. When experts were reminded that data were sought at the population level, consensus was attained, especially where qualitative choices were concerned. Where decisions were related to numeric categories (e.g. home range < 1 ha or 1-5 ha) there was greater uncertainty, reflecting the inadequacy of precise information. Nevertheless responses were consistently at the lower or upper ends of the scale for the same species.

Data were unavailable or very limited for some fields, such as specific information on "seasonal intolerance to fire" and "research projects undertaken". These fields were not included in analyses, but were retained in the database with view to capturing further information.

The Bradstock classification predicted 15 terrestrial bird species and 4 mammal species to be more sensitive than others in their class (Table 5.3). These species were all poor dispersers and take shelter in trees or in ground cover that is readily burnt. Food and nesting are affected if unburnt patches are not available post fire. The mammal and reptile groups were notable for the large number of **RpRtSt** species. These species are able to escape fire using permanent or temporary refugia, but are vulnerable to loss of ground cover and require an unknown combination of burnt and unburnt ground. Most bats were in this category. They are not generally affected directly by fire, but by reduced foraging opportunities, causing bats to move between burnt and unburnt habitat after fire (Chris Clague, pers. comm. 2005). Only reptiles and amphibians had **RtSpSt** species. These were tree dwellers such as frilled lizards, common tree snakes, and tree frogs that sheltered in trees but could forage in a range of post-fire conditions. All classes except amphibians had a reasonable number of generalists, **RpRtSpSt** species. These species are usually found in more than one habitat type and are able to tolerate a wide range of conditions. Amphibians are restricted to wetland and riparian areas, and the strength of the wet season affects populations more than the fire cycle.

Table 5.3 Using the 'Bradstock' classification, 15 terrestrial bird species and 4 mammal species were predicted to be more sensitive to fire than others in their class.

	Bradstock *et al.* classification								
	RpRtSpSt	**RpRtSp**	**RpRtSt**	**RpSpSt**	**RtSpSt**	**RpSp**	**RpSt**	**RtSp**	**RtSt**
Number of bird species	118	1	6	1	0	2	2	3	15
%	80.3	0.7	4.1	0.7	0.0	1.4	1.4	2.0	10.2
Number of mammal species	12	0	33	1	0	1	1	0	4
%	23.1	0.0	63.5	1.9	0.0	1.9	1.9	0.0	7.7
Number of reptile species	55	0	20	6	8	0	0	0	0
%	58.5	0.0	21.3	6.4	8.5	0.0	0.0	0.0	0.0
Number of amphibian species	0	13	0	0	4	0	0	0	0
%	0	76.5	0.0	0.0	23.5	0	0	0	0

The Lambeck classification aimed to identify focal species but there was inadequate discrimination between groups. The four classification options were too broad and, since the bioregion is only 0.3% cleared (NLWRA 2001), mobile species simply move to another suitable area. Most species were limited by factors other than fire.

The concept of increaser and decreaser species was only tested for birds. It was predicted that some species would decline to extinction with short fire return intervals, but others would decline if fire was completely excluded (Figure 5.4). An intermediate fire return interval would benefit some species more than others, but there would be no fire-induced extinction.

Figure 5.4 Number of bird species predicted to persist (black bars), decline (grey bars), or exist at optimal levels (white bars) given different fire return intervals.

5.3.2 Predicted Versus Actual Bird Responses

Paired t-tests showed that there was no significant difference in the number of bird species at burnt and unburnt sites ($p > 0.9$) when combining morning and afternoon surveys. However, there were significantly more birds at burnt sites than unburnt sites during morning surveys ($p < 0.05$), but these differences in abundance were not apparent during afternoon surveys ($p > 0.38$).

There was no clear separation between burnt and unburnt sites when presence/absence or abundance data were used in ordination. However, slight clustering between sites at the same property was discerned using presence/absence data (Figure 5.5). This may reflect inter-regional variation regarding species distributions. When each property was considered separately, burnt and unburnt sites were clearly different at Roxmere (Figure 5.6) and Bushy Park (Figure 5.7) but not at Calton Hills (Figure 5.8). Abundance data grouped by fire scar showed some separation between burnt and unburnt areas (Figure 5.9).

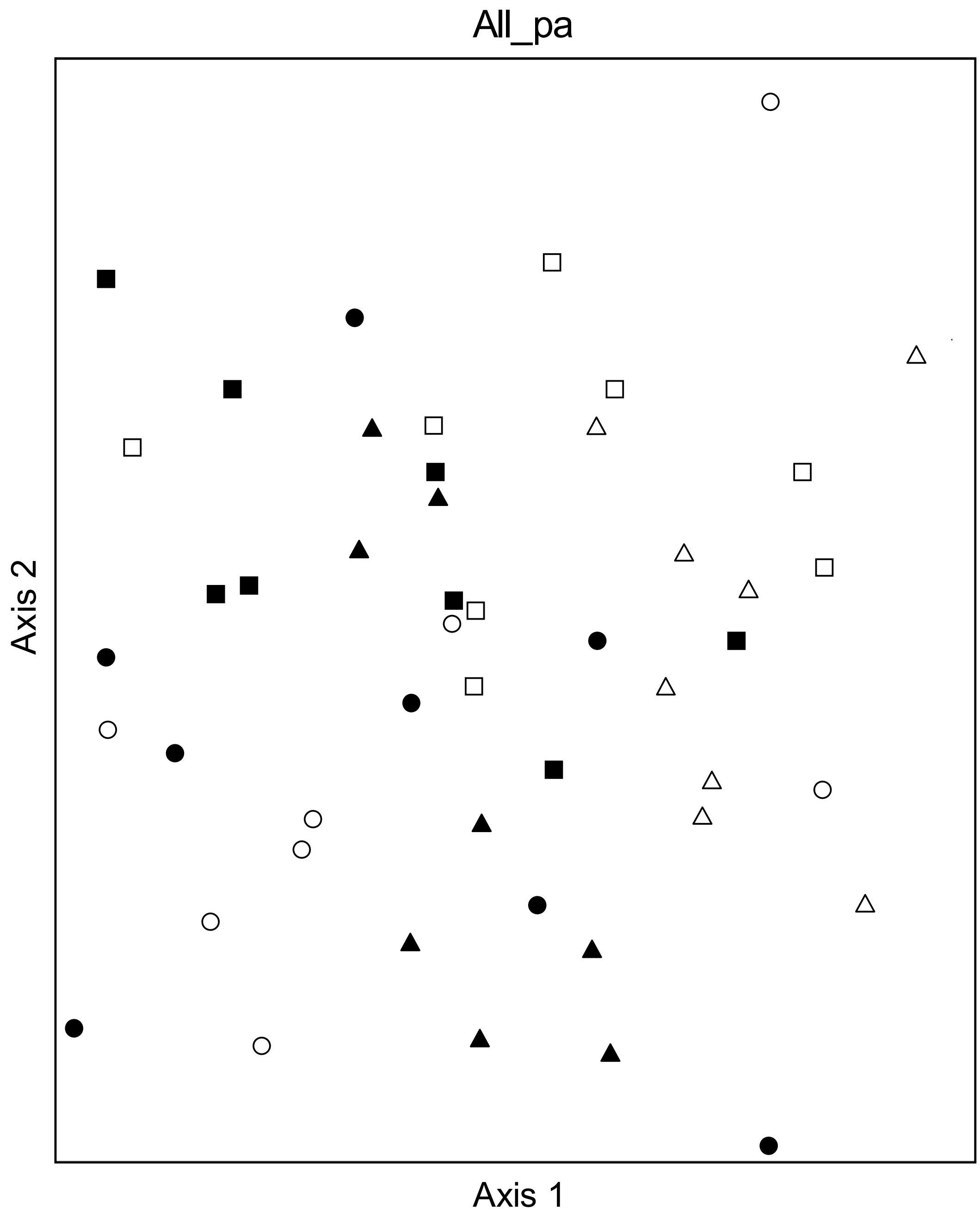

Figure 5.5 **Ordination based on the bird species sighted revealed slight clustering at the property level but not between burnt and unburnt sites. Circles = Roxmere, Squares = Bushy Park, Triangles = Carlton Hills, Black = burnt sites and white = unburnt sites.**

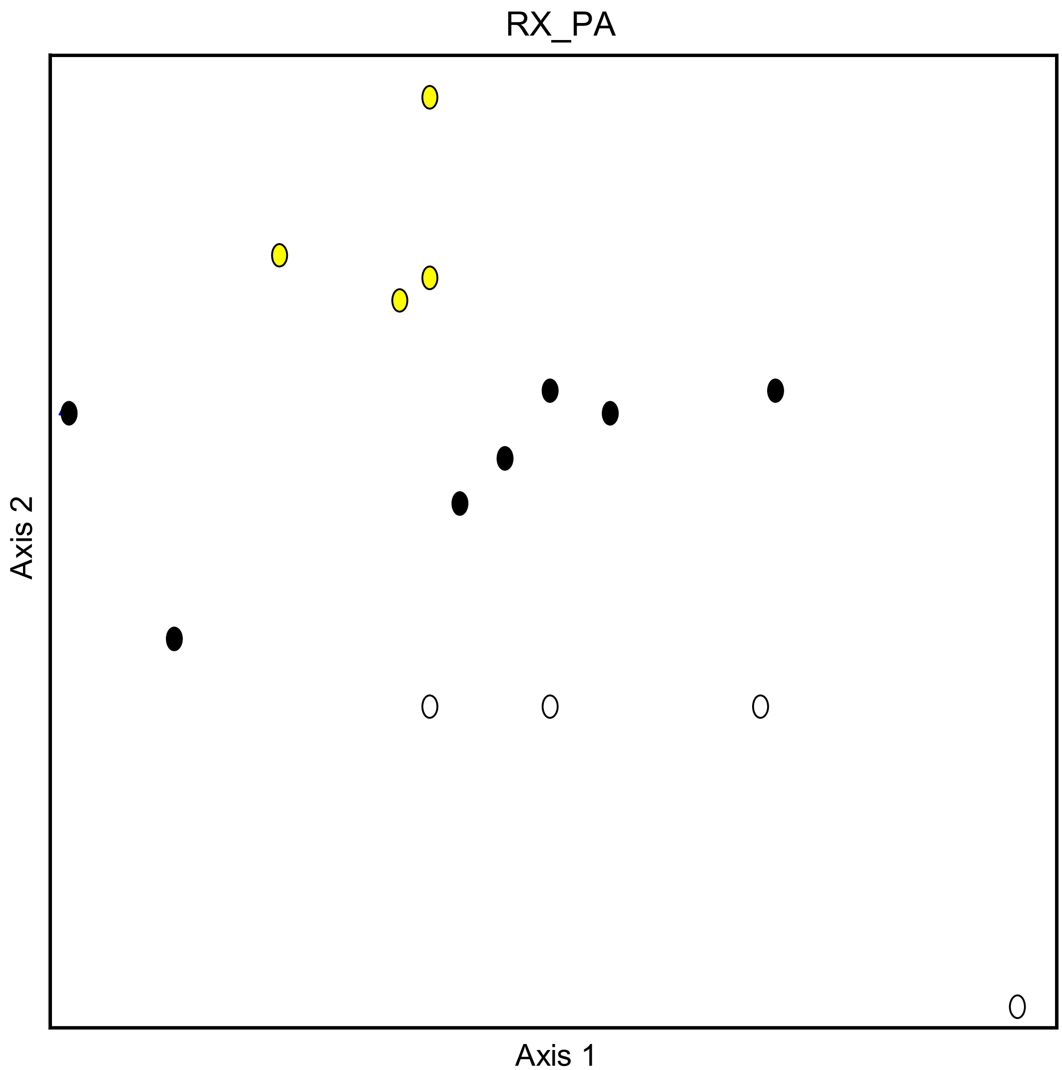

Figure 5.6 Based on the bird species present, burnt and unburnt sites at Roxmere could be discriminated from one another. However, the two unburnt sites were also different from one another. Black = Burnt, white = unburnt site (a) and yellow = unburnt site (b).

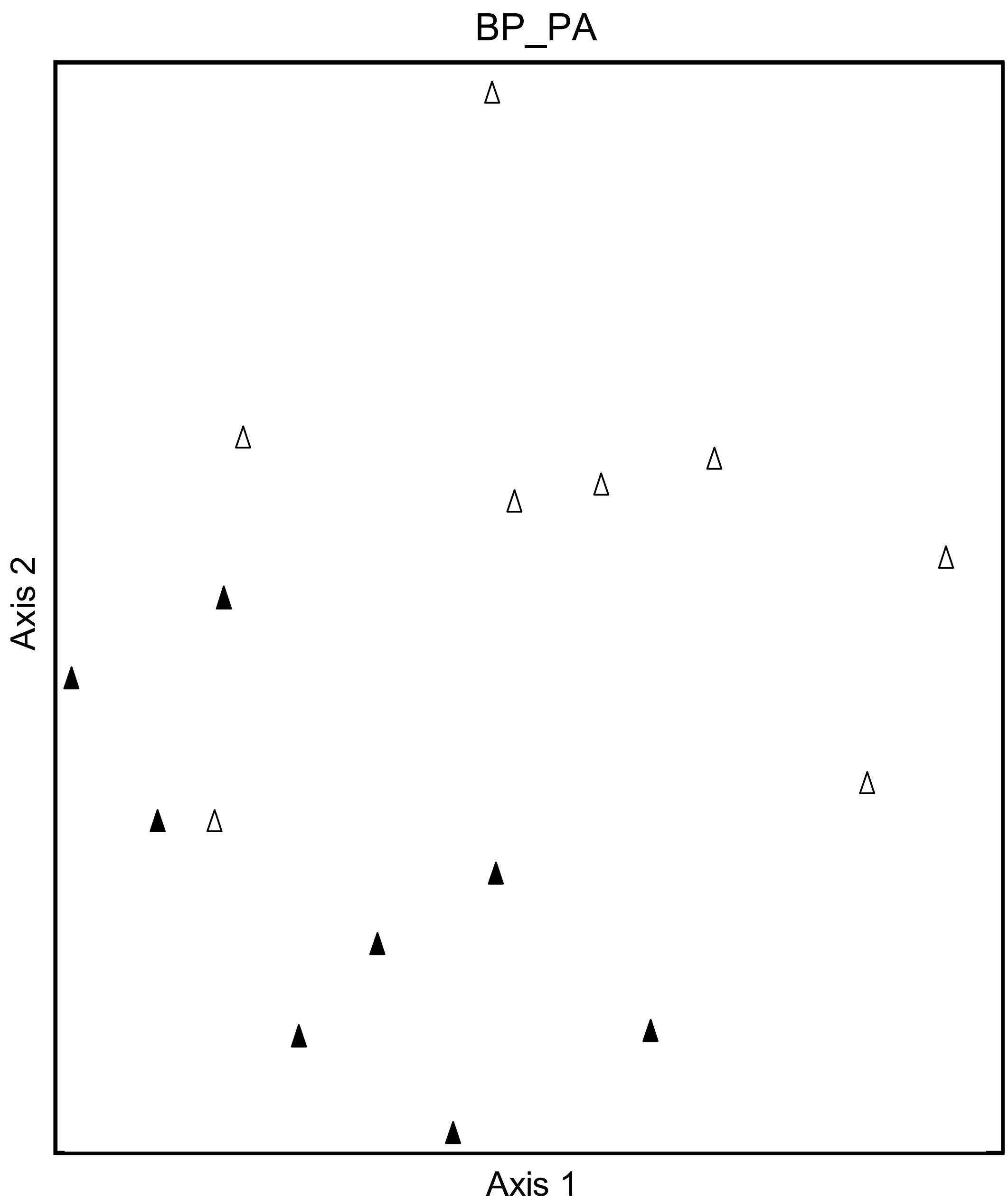

Figure 5.7 Based on the bird species present, most burnt and unburnt sites at Bushy Park could be discriminated from one another. Black = burnt sites and white = unburnt sites.

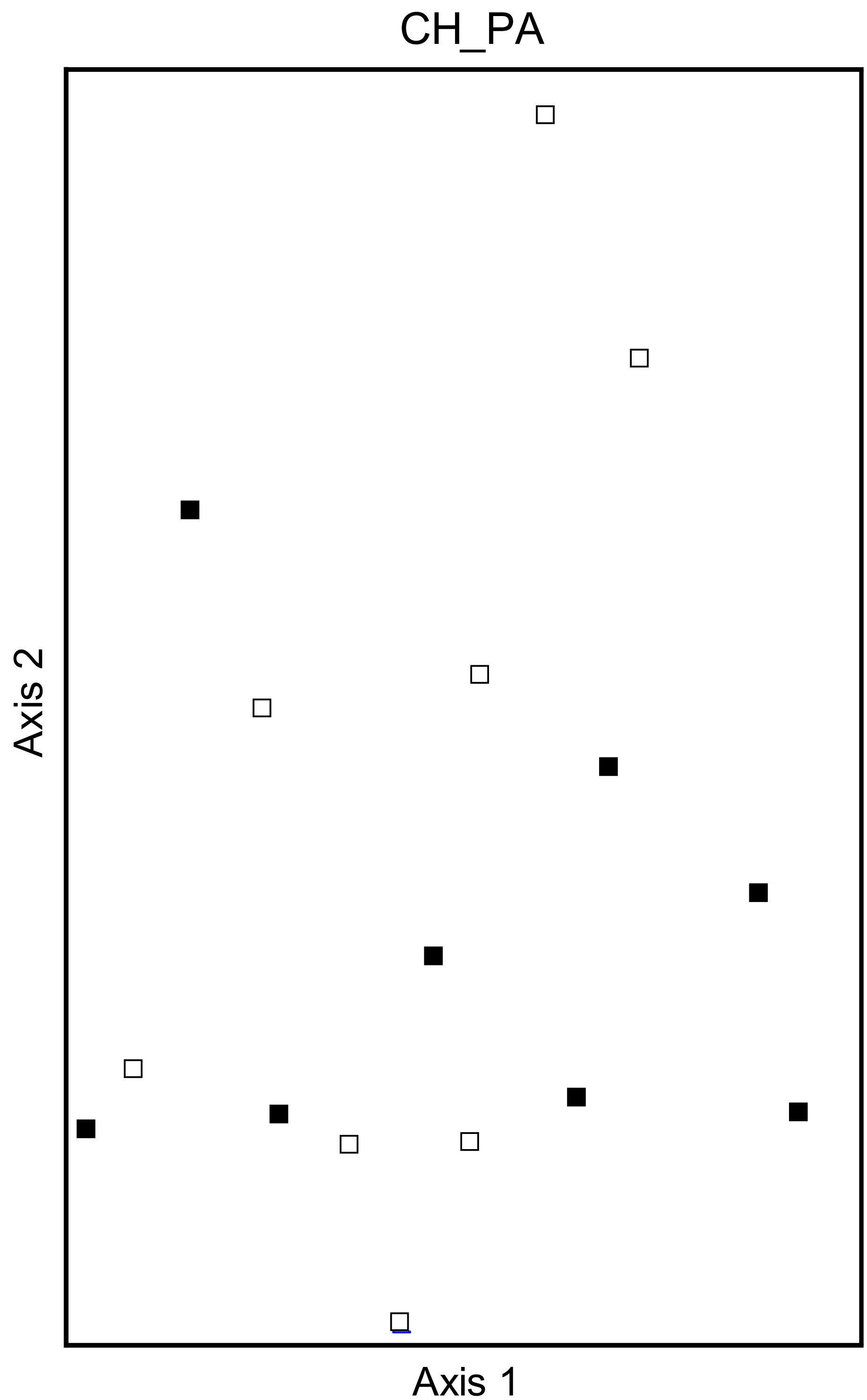

Figure 5.8 There was no clear distinction between the bird species recorded on burnt and unburnt sites at Calton Hills. Black squares = burnt and white squares = unburnt.

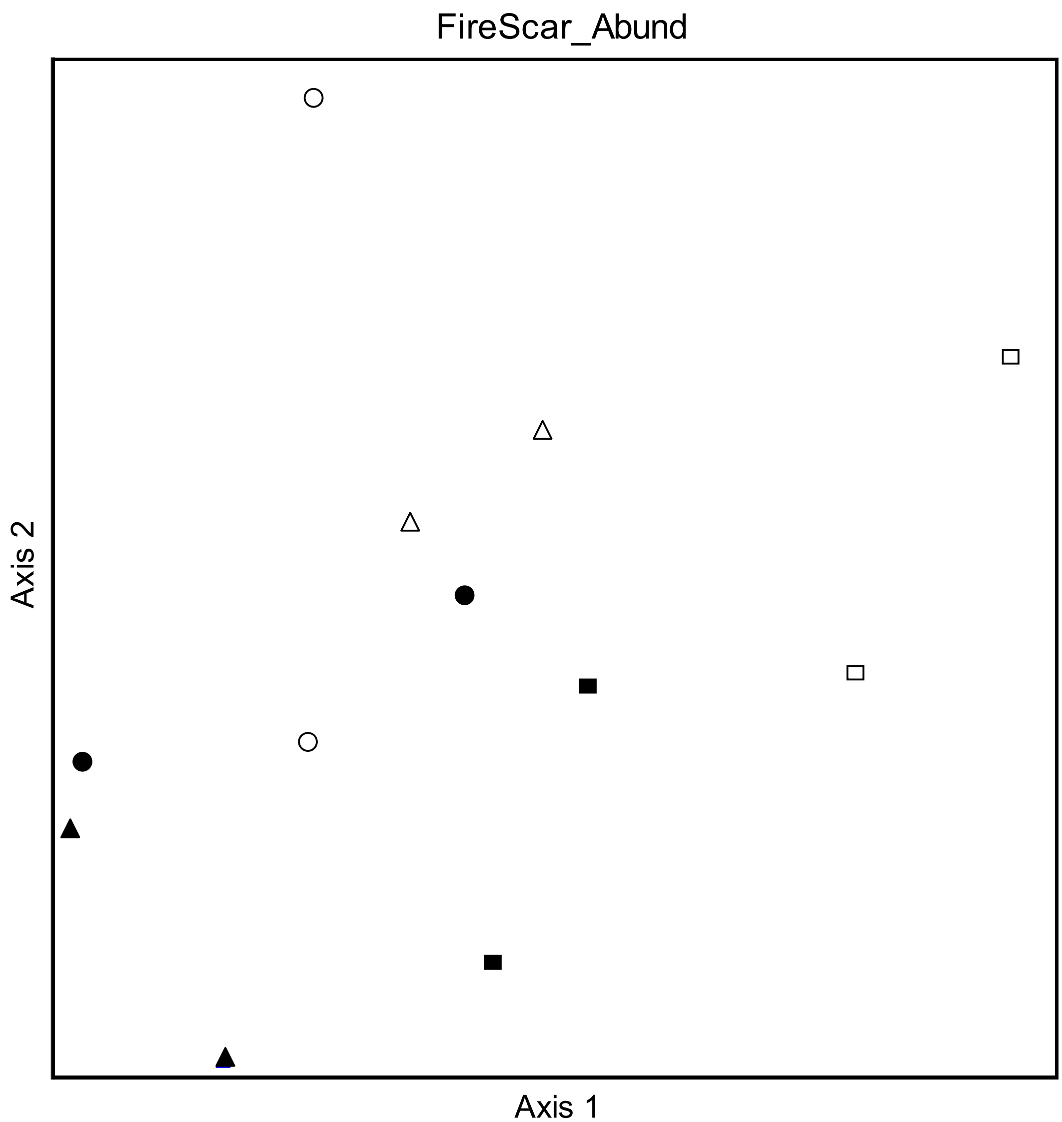

Figure 5.9 There were differences in bird abundance in burnt and unburnt vegetation when data from all fire scars were combined. Black = burnt, white = unburnt, circles = Roxmere, squares = Bushy Hills and triangles = Carlton Hills.

Only 6 of the 15 terrestrial bird species predicted as **RtSt** (the most fire sensitive grouping) were detected during the surveys (Table 5.4). Two species from the next most sensitive group, **RtSp,** were also detected. Chi-square tests confirmed that there were significantly more sightings of Variegated Fairywrens ($p < 0.06$) and Spinifexbirds ($p < 0.01$) at unburnt sites. These relationships were even stronger when data were regrouped so sightings at unburnt patches within fire scars were included with 'unburnt' records ($p < 0.001$ for both species). This approach showed that although Kalkadoon Grasswrens had been recorded equally at burnt and unburnt sites, they were 100% associated with unburnt spinifex patches within patchily burnt areas ($p < 0.001$). As predicted, these three species were strongly associated with unburnt habitat, either as patches within a fire scar or as entirely unburnt areas. The other species predicted to be fire sensitive were not detected in sufficient numbers for significance testing.

Indicator species analysis was conducted after removing records of species sighted fewer than three times, to concentrate on the more common species. Analysis was at the site level or scale, disregarding whether species were associated with unburnt patches within fire scars. Indicator values and their significance (Table 5.5) show that Spinifexbirds and Variegated Fairywrens are good indicators of unburnt habitat ($p < 0.05$). Other potential indicators were Magpies and Mistletoe birds ($p < 0.1$). Crested Pigeons, Crimson Chats, Diamond Doves, Willie Wagtails and Zebra Finches, all **RpRtSpSt** species, were strong indicators of burnt habitat ($p < 0.05$), followed by Budgerigars, Crested Bellbirds and Jacky Winters ($p < 0.1$). Magpies and Mistletoebirds were both classified **RpRtSpSt**, so their association with unburnt sites was unexpected. Chi-square tests supported this finding, with significantly more sightings of these two species at unburnt sites ($p < 0.05$). There were insufficient sightings on burnt sites to determine whether they were associated with unburnt patches.

Table 5.4 Fewer than half of the bird species predicted to be fire sensitive were sighted. The three species seen in sufficient number for statistical analysis (asterisk) were all associated with unburnt spinifex.

Predicted as fire sensitive and sighted			
Species	Classification	Number sighted at unburnt sites	Number sighted at burnt sites
Spinifexbird	RtSt	*11	1
Kalkadoon Grasswren	RtSt	*10	10
Carpentarian Grasswren	RtSt	5	0
White-winged Fairywren	RtSt	5	0
Little Button-quail	RtSt	2	2
Golden-headed Cisticola	RtSt	1	0
Variegated Fairywren	RtSp	*99	48
Brown Songlark	RtSp	0	1
Predicted as fire sensitive but not sighted			
Pheasant Coucal	RtSt		
Purple-crowned Fairy-wren	RtSt		
Rufous-crowned Emu-wren	RtSt		
Little Grassbird	RtSt		
Clamorous Reed-warbler	RtSt		
Singing Bushlark	RtSt		
Stubble Quail	RtSt		
Brown Quail	RtSt		
Red-chested Button-quail	RtSt		
Red-backed Fairywren	RtSp		

Table 5.5 Indicator species analysis was used to identify species with a strong association with either unburnt (grey shading) or burnt (no shading) vegetation. Species are ranked according to their calculated indicator value (IV).

Common name	Species	Observed indicator value (IV)	IV from randomised groups	Standard Deviation	p *
Zebra Finch	Sp77	62.2	48.5	4.67	**0.005**
Crimson Chat	Sp19	71.8	35.7	7.67	**0.001**
Variegated Fairywren	Sp67	44.9	34	5.75	***0.052***
Budgerigar	Sp12	39	28.2	6.16	***0.065***
Willy Wagtail	Sp74	47.9	26.6	5.61	**0.004**
Diamond Dove	Sp20	40.5	20.3	6.36	**0.007**
Crested Pigeon	Sp18	33.4	19.1	5.48	***0.025***
Jacky Winter	Sp31	28.4	18	5.46	***0.079***
Spinifexbird	Sp52	30.6	15.1	4.75	***0.021***
Australian Magpie	Sp36	24.3	15	4.69	***0.081***
Crested Bellbird	Sp17	22.7	12.7	4.68	***0.082***
Mistletoebird	Sp37	16.7	8.3	3.61	***0.1***
(For all other species p > 0.1)					

5.3.3 Patchiness Requirements of Indicator Species

Visiting eleven different fire scars to characterise species' patchiness requirements demonstrated that more of the target species were detected in the unburnt trial areas than within fire scars (Figure 5.10). No White-winged Fairywrens were sighted.

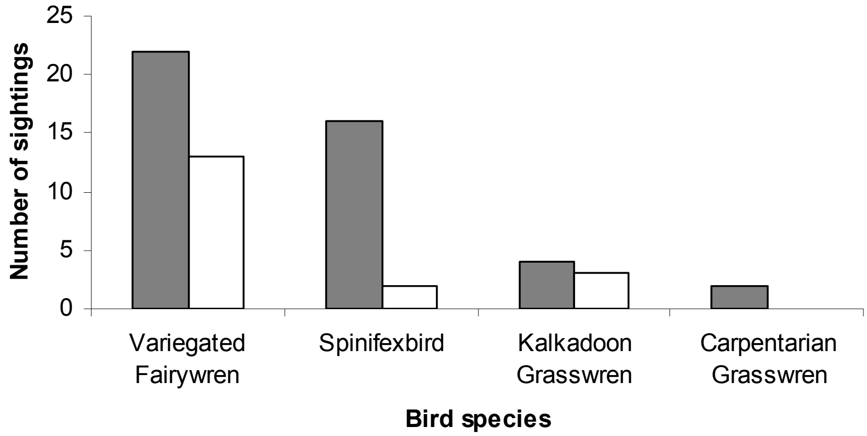

Figure 5.10 Indicator species were sighted more frequently during surveys of unburnt habitat (grey bars) than within fire scars (white bars).

Bird calls were played at 164 patches ranging in size from 1 m^2 – 10 ha (Figure 1.11). All but one of the 13 Variegated Fairywren sightings were on patches of spinifex between 50 m^2 and 5 ha in size. On these occasions there was another unburnt patch or area within 50 m. The spinifex was approximately 40 cm high and covered over 50% of the soil surface. In one instance Variegated Fairywrens were sighted in a nearby, resprouting shrub, not an unburnt patch. Spinifexbirds were only sighted twice; one was called from nearby unburnt habitat, the other was in a small patch of spinifex about 50 m from unburnt habitat. There were many other unburnt patches within a 50 m radius. Kalkadoon Grasswrens were sighted within fire scars 4 times: twice near the edge of unburnt areas and twice in rocky areas with a relatively small amount of thick spinifex. These data were insufficient for characterising patch requirements. However, the method provided significant insights to contribute to fire planning and monitoring in the Mount Isa Inlier. These insights are expanded in the discussion below.

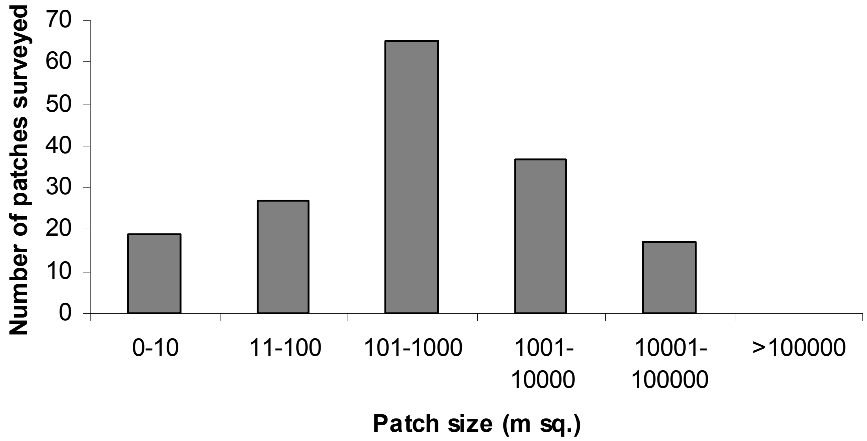

Figure 5.11 The distribution of patch sizes of unburnt spinifex within fire scars where the playback technique was used to survey target bird species.

5.4 Discussion

The structured, systematic approach used in this study provided a means to translate ecological principles into applied management. From the premise that fire patchiness is important, preliminary recommendations can now be made on a desirable scale for mosaic burning in the Mount Isa Inlier. The fauna fire-response database gave a starting point for allocating species to functional groups. The categorisation process enabled the multitude of fire responses to be sieved and integrated and was used to identify fire sensitive species. The strong association of these species with unburnt spinifex was confirmed, but ideal characteristics for unburnt patches within fire scars could not be defined. From the insights obtained, however, it is suggested that fire size should be restricted to 100 ha (or thereabouts) in this bioregion. The rationale for this target is explained below. The work is significant because the method allows for transparent objective setting for ecological fire management. The result is a quantifiable target that can be monitored using readily available remote sensing techniques, as well as guidance on a suite of indicator species that could be used for associated ecological monitoring. The following discussion explores issues raised by the methodology and makes recommendations on how it could be refined.

5.4.1 The Fauna Fire-Response Database

The fauna database considered issues to do with time, space, the effects of individual fires as well as a sequence of fire events. It addressed direct impacts of fire (e.g. vulnerable at nesting time), as well as indirect affects (e.g. habitat change over time). Data were captured on the full range of possible fire responses in order to explore different functional group classifications for simplification and prediction. This comprehensive approach facilitated discussion of different fire management scenarios with experts, but raised issues related to terminology, database structure and inconsistent species' responses. For example, the 'immediate response to fire' can be different for birds that are nesting, and 'response after fire' requires that the post-fire timeframe be clearly defined. There was some ambiguity where time and space criteria were mixed within the same menu option. These issues were addressed by providing written instructions and clarification to experts, but these issues may have been circumvented by more thorough testing before data collection commenced (Burgman 2005). Linguistic uncertainty is inevitable where there is a continuum of possible responses since responses are always context specific and many words have more than one meaning (Burgman 2004). Overall, the database provided a framework that ensured the majority of terrestrial vertebrate species could be objectively considered. It also drew attention to specific issues that may have been overlooked. For instance, it might be assumed that amphibians are protected from fire because they dwell in moist riparian areas. But frogs that shelter in trees can be affected by a series of wildfires that destroy riparian habitat. Similarly, bats disperse by flying but depend on tree hollows that may be destroyed by frequent fire.

Expert opinion provided an efficient way of synthesising a broad range of knowledge, especially when it was difficult or impossible to acquire data directly (Burgman 2004, 2005). There are many aspects to consider. Data reliability depends on how the experts are interrogated, their experience, ability to recollect accurately, optimism, self-confidence and motivation. To minimise subjectivity, Burgman (2005) recommended selecting a panel of experts according to defined criteria. Further, a well-planned, explicit process can minimise bias and improve consensus decision-making. Many of the suggested procedures were used here, but the study would benefit from broader consultation across the vertebrate classes and from face-to-face group discussions to resolve differences of opinion.

5.4.2 The Application of Functional Groups

Fire-sensitive species were readily identified using the functional group classification proposed by Bradstock *et al*. (2005). It was surmised that species relying on transient refugia and requiring vegetation in a particular post-fire condition would need a certain level of internal fire patchiness in order to survive. The fire sensitivity of these species would be greater than for species that tolerate a wide range of conditions. The classification system worked well for separating the two extremes, robust **RpRtSpSt** species or sensitive **RtSt** species. It also catered for species with an intermediate response, such as tree snakes. These snakes shelter in trees during fire but can hunt in vegetation at any time after fire (**RtSpSt**).

The Lambeck classification was not as useful. Identification of ‘focal species’ was recommended for prioritising conservation actions in fragmented landscapes (Lambeck 1999). If dispersal is an issue, remnant vegetation can be linked by revegetating corridors; if species are limited by area, patches can be enlarged by tree planting; and if species are resource-limited, favoured species can be planted for supplementation. Considering area, dispersal and resource limitations for fire management was conceptually appealing, but the method was difficult to apply. The landscape of the Mount Isa Inlier is relatively intact so mobile species can move to suitable habitat. Individuals of sedentary species might be killed, but populations and meta-populations can survive at the regional scale. Also, many species are limited by factors other than fire, such as the strength of the wet season.

Classification according to whether a species' population would increase or decrease with recurrent fire enabled speculation on the response to different fire frequencies. It assisted with evaluating management options but did not provide information on patchiness requirements. Findings were similar to other fire prone areas, where frequent, repeated burning and total fire exclusion are problematic for many species and can lead to the eventual decline of these species (Mills 2004; Tasker & Baker 2005). Therefore, fire in the spinifex landscape is beneficial, at some unknown, intermediate fire frequency.

The 'general impression' category allowed experts to offer an opinion based on first principles rather than direct knowledge. Although subjective, it provides a starting point for further observation and testing. A 'best bet' satisfied experts' desire to contribute their view and this was considered better than no data. It highlighted an important issue: the data supplied were relevant to the Mount Isa Inlier but would need to be reviewed for use in different bioregions. For some species the 'general impression' was related to vegetation recovery after fire, which may differ between environments (Keith MacDonald pers. comm. 2005).

5.4.3 Fire Sensitive Species: Predicted Versus Actual

Bird species predicted as being fire sensitive using the Bradstock classification (Variegated Fairywren, Spinifexbird, and Kalkadoon Grasswren) did favour unburnt areas. However, species such as the Carpentarian Grasswren, White-winged Fairywren, Little Button Quail, Golden-headed Cisticola, and Brown Songlark were not seen in sufficient numbers for statistical analysis. Some fire-sensitive species were not sighted at all. Birds such as the Purple-crowned Fairywren, Singing Bushlark and Clamorous Reed Warbler are not typical of spinifex habitat, although they are present in the bioregion. Other **RtSt** and **RtSp** species may have escaped detection or were absent during the season and time available for survey. These issues are inevitable in short-term rangeland surveys (Mac Nally *et al*. 2004).

Indicator species analysis suggested two additional species that might be fire sensitive. Mistletoebirds and Magpies were sighted significantly more often on unburnt sites. Neither of these species was identified as sensitive by the functional group classification because they are both good dispersers and occupy a wide range of habitats. Mistletoebirds may be negatively affected by fires that scorch tree canopies, however. Such fires would kill the mistletoe plants upon which these birds are dependent. This fire sensitivity is not necessarily a cause for concern given the wide distribution of Mistletoebirds. There is no clear explanation why the broadly distributed, omnivorous, and apparently fire tolerant Australian Magpie was recorded more frequently at unburnt sites.

Woinarski (1999) suggested that minor changes to the fire regime might be as critical as catastrophic wildfires with regard to bird decline. Imperceptibly, gradual habitat change can result in species loss in the long term. Using this rationale, the functional group classification overlooked species affected by fire operating at a more pervasive level. For example, too-frequent fire may result in the eventual decline of hollow nesting birds. These ‘fire sensitive’ species were classified as **RpRtSpSt** as they are good dispersers and occupy multiple habitats and different stages of vegetation development. It is interesting to note that hollow nesting birds declined in the Mount Isa Inlier between the 1981 and 2002 compilations of the Australian Bird Atlas (Stephen Garnett pers. comm. 2005).

The effectiveness of Bradstock and others functional group classification should be tested further by more extensive sampling, both spatially and temporally.

A number of issues regarding bird survey techniques need to be considered, however (see Watson 2003 and Mac Nally *et al*. 2004). For the burnt/unburnt comparisons, I addressed the issue of patch size by choosing the six largest fire scars available. The influence of edge effects on the data was minimised by commencing surveys at standard distances inside and outside burnt areas. Variation in vegetation was minimised by restricting sites to spinifex/snappy gum communities. Aspects such as fine scale habitat complexity, distance from water, internal fire patchiness, grazing intensity and rainfall since fire were not addressed. These are known to be important but standardisation is nigh impossible unless an experimental approach can be used.

The fire responses detected in the bird fauna support findings from other savanna areas. Most birds are highly mobile and able to respond rapidly to changes in habitat and food availability. Changes in the bird community are more subtle than in temperate areas where there are clear links between the species present and stage of vegetation development (Braithwaite 1996; Woinarski 1990). The general pattern is for species of functional raptors (carnivores and carrion eaters) to be attracted to fire, with nomadic open-country species moving in soon after (Woinarski 1999). Recently burnt areas attract seed-eating birds as well as insectivores that feed on insects drawn to new plant growth. Thus, species like Budgerigars, Zebra Finches, Diamond Doves, Crested Pigeons and Crimson Chats are common after fire (Woinarski 2005). The number of species may not differ between treatments (Mills 2004), but bird abundance can be greater in the months following fire (Braithwaite 1996; Christensen & Kimber 1975). Species that require unburnt habitat or a particular burnt/unburnt combination may take time to recolonise. Regional scale movements also occur as some species track resources from one recently burnt area to another (Woinarski *et al*. 2005).

It was advantageous to have data centrally collated so management options could be systematically evaluated. Birds may not be the most sensitive group, due to their mobility (Mac Nally *et al*. 2004). The alternative view is that most birds require trees, which might recover more slowly than ground cover. If so, ground-dwelling mammals, reptiles and amphibians would recover more quickly (Woinarski 1999). Birds were the most practical group for rapid assessment, however, which was important in the short time available for this study.

Overall, the Bradstock classification was useful to identify bird species likely to have special patchiness requirements. Further exploration of the concept is required. The key advantage is that it enables transparent setting of objectives. Subsequent fire management can then be assessed by monitoring the presence or absence of fire-sensitive species. In other areas, functional groups could be based on preferred vegetation state because bird communities of the ground and shrub layers show more pronounced changes after fire than tree-canopy dwellers (Recher & Christensen 1981). This is unlikely to work in the semi-arid savanna where there is less height separation between strata in low open woodlands. Also, ‘cool, patchy’ spinifex fires are difficult to achieve and tree canopies are frequently defoliated. Consideration of refugia requirements is more appropriate in the Mount Isa Inlier.

5.4.4 Determining Patchiness Requirements

Determining the patchiness requirements for key species was not straight-forward but provided significant insight. Importantly, the results supported the notion that managing for fire size, rather than a level of internal fire patchiness, would cater for fire sensitive bird species in the Mount Isa Inlier. Application of the methods, rather than the actual data, suggested that a fire size of around 100 ha could be used as an initial target. This is based on the following:

- Survey results confirmed the fire sensitivity of **RtSt** species. These species were loyal to unburnt habitat and, if present within a fire scar, they were entirely associated with unburnt patches of spinifex.

- I intended to survey 20 fire scars to evaluate patchiness requirements. However, only 11 were large enough to survey. The other nine were inspected prior to survey but it was obvious that the playback technique would simply call birds in from adjoining unburnt habitat. At one fire scar sampled, the survey was halted early to prevent recorders sampling the same area. Thus, 'small' fires seem to be achievable.
- The three recorders surveyed approximately 48 ha per fire scar. It was necessary to be well within a fire scar to prevent birds being called in from the edges. At less than 80 – 100 ha, the influence of the fire scar edge overwhelmed the response. Around 100 ha was considered the minimum fire scar size for the sampling technique. This ensured an adequate core area for detecting change.
- It was hypothesized that fires of this size would affect individuals but have virtually no effect on the meta-population. Fires less than 100 ha ('too small to sample') would also be suitable as species could remain *in situ.*

One hundred hectares is promoted as an initial 'best guess' as an acceptable fire size. It allows entry to the adaptive management cycle by providing a measurable target. The outcome of management actions can be readily monitored using existing remote sensing techniques. This fire size may not be 'the solution', but provides an initial long term goal and steers management towards a reduction in the current mean area burnt.

'Patchy' fires might be conceptually desirable but are difficult to plan for in spinifex habitats. Unburnt patches within fire scars are nonetheless assured by inherent landscape variability. Although different species respond to different scales of patchiness and no one species is an ideal indicator (Mac Nally *et al*. 2004), results here provide a starting point. The fauna fire-response database could be consulted regarding species mobility and home range for further guidance.

Studies to determine specific internal patchiness requirements would need to address a number of issues. It was sometimes difficult to ascertain whether areas of unburnt spinifex were isolated within the fire scar or contiguous with the 'unburnt area'. At what size should very large unburnt patches be classified as 'unburnt area'? Prior definition of patches using remote sensing may assist, but this requires imagery at an appropriate scale and substantial analysis before undertaking field work. Also, internal fire patchiness is much greater in reality than it appears on imagery (Price *et al.* 2003). This may not affect bird surveys because, when target species occurred within fire scars, they were associated with relatively large patches of unburnt spinifex. Small patches, say 10 m x 50 m or less, seemed to be irrelevant. Fire history is important too. In this case, if the sample area had burnt within the last 5 years, target species may not have recolonised. Finally, sampling techniques need to be reliable. Birds may not have responded consistently to calls. Although Kalkadoon Grasswrens unfailingly responded within 2 minutes of playing the call, on one occasion a Spinifexbird was sighted prior to play-back, but would not reappear.

5.4.5 Management Implications

At least one of the targeted fire sensitive bird species was detected outside a fire scar before surveying its interior. It could be surmised, therefore, that **RtSt** species were being catered for under the current fire regime. A review of fire size data (Chapter 2) suggested otherwise. Only 1% of fires were under 100 ha and 3% were under 200 ha. The average fire size over a six year period was approximately 6,200 ha, although this fire size was skewed by a few large wildfires. The scale of imagery may not have been suitable for recording the many small fires that occur, but it is obvious that the large, homogenising fires need to be avoided. A relatively small proportion of the landscape burns each year compared to mesic savanna areas

(Felderhof & Gillieson 2006). However, the distribution of fire size needs to be considered. These data suggest that changes towards a greater number of small fires would be beneficial.

The second outcome from this study is guidance for ecological monitoring at the property or regional scale. The functional group classification identified a suite of fire sensitive species and these could be scored for presence/absence at the end of each fire season. **RtSt** species suggest a certain level of landscape patchiness, so their presence would help gauge whether fire size and internal patchiness were appropriate. Being able to choose from a group of species addresses the issue of rarity and overcomes the need to account for specific habitat preferences. A tailored search effort might be necessary, however, if particular species have not been sighted for some time.

5.4.6 Implications for Research

Translating ecological principles into applied management requires further work. First, the fire-response database should be refined by reviewing the various fields and menu options. Amendments to clarify ambiguity, or modify terminology to ensure no redundancy, would improve data capture. Then, additional experts should be consulted. A workshop approach that enables debate but aims to reach consensus is desirable. The resulting guidelines would assist planning and provide a framework for collating additional information. Inevitably, a large number of 'unknowns' would remain. The database could then be used to guide research priorities. For example, research attention could be directed to autoecological studies that concentrated on **RtSt** species.

Identifying key indicator species may aid Australian adoption of the 'Thresholds of Potential Concern' (TPC) approach used in South African fire management (Parr & Andersen 2006). This approach sets targets for the seasonal distribution of fire and the amount of area to be burnt. Links between biological monitoring and fire records are just commencing. The fire-response database/functional group classification might help in choosing which taxa to focus on, but research is required to determine how this might work, especially in setting lower population thresholds.

Further testing of acceptable fire sizes for different taxa, concentrating on **RtSt** species, is required. Fire size and internal fire patchiness are issues of scale; an appropriate fire size for mobile species such as birds may not be suitable for less mobile species. The procedure used here has potential for rapid assessment. If an area is too small to sample, it could be speculated that the fire had little effect on the species' population.

The Lambeck classification was not as useful as the Bradstock classification in this study, which highlighted that approaches to biodiversity planning based on fragmented landscapes, with either/or choices, cannot be readily adapted to more natural landscapes such as those in the Mount Isa Inlier. This concern extends to population models with either/or assumptions (Bascompte & Sole 1996; With & King 1999). Such models are unlikely to assist conservation in savanna landscapes which are relatively intact. Hobbs (2005) explained that these landscapes have been modified to varying extents and thus are functionally fragmented. He examined the differences and similarities between intensive and extensive land use areas and suggested how concepts based on fragmentation could inform savanna management. Further testing is necessary, but either way, knowledge of species specific responses is required (Hobbs 2005). Maintaining habitat is more cost effective than reconstruction, so effort should be directed to developing these links, as well as modelling species performance in continuous landscapes. Understanding responses to fire management is an obvious application.

Finally, I hypothesize that fire is required by some fire sensitive species to rejuvenate habitat. Species such as Kalkadoon Grasswrens, which are restricted to hummock grasslands near Mount Isa, present a challenge. They require long-unburnt spinifex, but the spinifex itself becomes moribund without occasional fire. Fires can eliminate individual animals from a fire scar; if adults can disperse, it may lead to genetic mixing which potentially benefits the meta-population. Alternatively, fire may increase the rate at which juveniles replace adults as juveniles establish in recovering habitat. Determining whether a mosaic of burnt and unburnt country is critical, and the optimum fire frequency, requires further research.

5.5 Conclusion

This chapter offers a way to put the patchiness paradigm into practice. A structured and systematic approach, in which the wide range of species responses are considered and then classified into a manageable number of fire-response categories, will assist to set management objectives. For the Mount Isa Inlier, a fire size around 100 ha is a reasonable target, especially given the inherent level of internal fire-patchiness. Thus, managers should aim to restrict fire size rather than aim for a level of internal patchiness that is difficult to specify, difficult to achieve, and difficult to monitor. This fire size can be readily monitored using existing remote sensing techniques (see http://www.firenorth.org.au). Ecological outcomes could be monitored by selecting a suite of fire sensitive species and assessing them for presence/absence at the property scale after each fire season. The key issue becomes the prevention of very large fires that homogenise the landscape. Establishing and maintaining a fire mosaic should assist to minimise fire spread, especially in the Mount Isa Inlier where fires reliably abut recently burnt areas. The ideal distribution of fire in the landscape needs to be determined, however. An intermediate level of burning, rather than no fire, is recommended. This has the potential to prevent wildfire and rejuvenate the landscape, but requires active fire management on an annual basis.

6. SUMMARY, FUTURE RESEARCH AND CONCLUSIONS

6.1 Thesis Objectives

This thesis investigated a number of issues related to fire ecology and management in the spinifex/snappy-gum woodlands of northwest Queensland, Australia. The aim was to inform ecological fire management because fire-related research is scant for the bioregion (Mount Isa Inlier) and other semi-arid parts of Australia's tropical savannas in general. There is little guidance for land managers wanting to adopt ecologically sustainable fire management practices. When, how often, and how much should they burn, and how should they evaluate success? The landscape 'patchiness paradigm' provides some direction. It suggests that numerous small fires, with variety in fire timing, frequency and intensity, will lead to habitat diversity across the landscape, benefiting species conservation. The paradigm, based on general ecological principles, encourages the implementation of fire regimes that develop and maintain fine-grained landscape patchiness. However it does not suggest an optimal fire size, level of internal fire patchiness or how to determine if species will be retained or lost. This lack of clear targets makes it impossible to measure success or failure. Outcomes are uncertain if the patchiness paradigm is directly applied. A more strategic approach would allow measurable fire management goals to be set and would facilitate the link between ecological theory and applied management actions. In this chapter I develop this link.

6.2 General Approach

The study used remote sensing technology to examine landscape fire patterns and to identify key drivers of these patterns. Field surveys were undertaken to determine regenerative responses of the vegetation and investigate spatial variability between and within fires. This gave an understanding of fire-mediated vegetation dynamics in spinifex/snappy gum communities. A fauna fire-response database was compiled using expert opinion. Different options were considered for classifying species according to their fire sensitivity. A classification related to species' assumed refugium and seral (vegetation maturation) stage requirements appeared to be useful. The predicted response of fauna was then tested in the field using birds as a sub-group. Bird species relying on long-unburnt spinifex during and after fire were more vulnerable to fire than generalist species. Additional field studies were undertaken to determine whether these bird species had specific internal fire patchiness requirements. The results were interpreted in terms of current fire patterns operating in the Mount Isa Inlier and the desirable level of landscape patchiness. Suggestions were made on how spatial fire patterns might be monitored using satellite imagery and ecologically using a suite of indicator species.

An hierarchical approach was taken based on advice that observations made at multiple scales are the key to unravelling complexity (Bissonette 1997). The 'hierarchy of scales' (Turner *et al.* 2001; Wessman 1992) is based on the tenet that landscape pattern can be discerned at nested levels of scale. There is no one 'correct' scale of investigation; conversely, cross-scale studies are recommended (e.g. Ludwig *et al.* 2006; Turner *et al*. 2001; Bisonnette 1997; Wessman 1992). Analysing patterns at a broad scale provides context and allows processes to be inferred, while fine-scale studies provide the details on the processes needed to explain patterns at higher levels.

Therefore, information was collected at broad scales using remotely sensed data and at fine scales by investigating the fire responses of different plant and animal species. The approach provided a useful framework for integrating data from different disciplines.

A second feature of the work was that sites were located broadly across a bioregion. Many fire studies are based on relatively small areas where plots are set up to examine a restricted subset of the potential fire regime, preferably in an experimental situation (Gill 1999). But fire is an ecological process operating at the landscape scale and an enormous range of fire intensities, frequencies and seasons are possible and typical. Fire management needs to be undertaken in this context. An understanding of ecosystem and landscape functioning, preferably in the 'real-world' setting, is an important first step for predicting the ecological significance of fire management outcomes. The bioregional approach engendered confidence in extrapolating from site-based measures to spinifex/snappy gum communities in general.

The adaptive management cycle was adopted as a framework for interpreting the results. Adaptive management is frequently recommended to land managers for making progress in the face of complexity and uncertainty (e.g. Holling 1978; Andersen 2003; ESA; Lindenmayer & Burgman 2005). Applying this planning cycle enabled links to be demonstrated between research findings and management practices. It was a convenient construct, since my research specifically aimed to inform fire-management planning and monitoring.

6.3 Summary of Results

6.3.1 Bioregional Patterns

Fire patterns in the semi-arid Mount Isa Inlier had greater affinity with arid Australian landscapes than mesic savannas. Like arid Australia, rainfall is low, spinifex is the dominant ground cover and fires can occur in any season. The amount of summer rain varies considerably from year to year affecting fuel growth. Extensive wildfires commonly occur after seasons of above average rainfall. Unlike arid areas in central Australia, wet-season rain **reliably** falls each year in the Mount Isa Inlier bioregion. This results in a staged response by the vegetation, with distinct growth flushes each wet season until there is sufficient fuel to sustain a subsequent fire.

Although the area burnt each year was highly variable, the fire-season and fire-frequency results were more consistent. The earlier prediction for the Mount Isa Inlier region was that the fire season would begin in late March and last until early January the following year (Walker 1981). This was based on estimated fuel dynamics. Walker hypothesised that plant growth would cease after the wet season and fuel would cure over the dry season, resulting in increased flammability. My findings document that the pattern of burning is strongly tempered by social norms. The fire season actually commenced around August and the area burnt increased dramatically until the onset of summer storms. The area burnt reduced as the wet season became established, but fires occurred reliably through to March. There were virtually no fires between April and July. This pattern reflected the desire of pastoralists to conserve grass as cattle fodder for as long as possible after the wet season. Pastoralists actively burn with the onset of summer storms to reduce woody-weeds (principally native *Acacia* species), promote new growth in spinifex, or to reduce fire hazard. Deliberate ignition continues over the wet season, although fires are also ignited by lightning. Thus, fires in the Mount Isa Inlier occurred in the ‘late-dry season’ and during the ‘wet season’.

Fire frequency in the central arid zone is partly determined by the regenerative capacity of spinifex (Allan & Southgate 2002). The fire return interval was 4-5 years for the Mount Isa Inlier, which compared favourably with semi-arid parts of the Northern Territory, Australia (Allan & Southgate 2002). Approximately 50% of the landscape remained unburnt over the six year period studied.

The majority of fires were less than 100 km^2 in size and 96.5% were less than 300 km^2. Very large fires were the exception, but did occur. The largest fire recorded during the 6-year study period was 1,500 km^2. This was considerably smaller than the 10,000 km^2 fires described for spinifex country in arid central Australia (Allan & Southgate 2002). The query arose whether fire-generated patches in the Mount Isa Inlier were ‘small’ enough? If so, has the generic goal of developing and maintaining fine-grained landscape patchiness been attained?

6.3.2 Vegetation Dynamics

The pathway of vegetation change in spinifex/snappy-gum communities was highly predictable, with post-fire changes typical of those described for arid spinifex communities. Burning completely removed most of the cover, but the vegetation was restored over time as species resprouted or were recruited from the soil seed bank. Species diversity peaked in the second year post fire and decreased over time. This was driven by changes in the ground cover. A flush of ephemeral species emerged in the immediate post-fire years but spinifex dominated the ground layer thereafter. The diversity of shrub and tree species did not change significantly post fire. Most perennial trees and shrubs remained in the community, although they were initially reduced in size. All perennial species, including the obligate seeding shrubs, had commenced flowering by the third year after fire. These demographic analyses helped with interpreting community dynamics, but all maturation stages were present in all strata, at all times since fire. Multi-aged stands and spatial variability within fire scars was also noted by Ooi *et al*. (2006), who highlighted the importance of this patchiness with respect to species persistence.

The system demonstrated resilience to fire. There was insufficient fuel for fire to spread for at least three years from the previous fire. By then, most ephemeral species have set seed, and perennial species have started to produce seeds. Also, the inherent variability within the burnt area meant that vegetation at early recovery stages co-existed with patches of vegetation at later recovery stages. Mature plants in the unburnt patches provide reproductive material that ensures species remain in the system.

6.3.3 Variation in Fire Severity

Fire does not act uniformly across a landscape due to variation in wind speed, topography, vegetation and time of burning (Turner *et al.* 2001). The large amount of variability within any one fire event is well recognised (e.g. Lindenmayer & Burgman 2005; Atkins & Hobbs 1995; Tolhurst 1995; Whelan 1995; Willams *et al*. 1994), but it is particularly notable in discontinuous hummock grasslands (Gill *et al.*1995). Leaf scorch, tree char height and shrub twig diameters were used in this study as indicators of fire severity and demonstrated that severity varied as much within fire scars as between fire scars. Differences in fire severity were detected between paired transects at almost half the sites, although this level of variability was often not visually apparent.

Understanding the inherent variability of fire within a landscape is important for developing the patchiness paradigm and for planning the use of fire to manage vegetation. Spatial variation or patchiness within fire scars will make the next fire more variable, leading to a cyclic pattern of heterogeneity. Importantly, few of the fire scars visited looked ‘patchy’; most appeared to have been intensely burnt. The differences were measurable nonetheless and fire scars could be plotted on a continuum of fire severity. This highlighted the difficulty in planning for a ‘patchy’ fire in spinifex/snappy gum communities. It appeared that once spinifex communities were ignited, they burnt at high intensity. When this is coupled with the natural unevenness in severity, it suggests that emphasis on internal fire characteristics may not be necessary. Patchiness will occur regardless. Thus managers can focus on achieving a target fire size as their management goal.

6.3.4 The Response of Fauna

The process adopted for incorporating the needs of fauna into fire-management planning could be applied to other geographic areas. The shortage of empirical studies was addressed by capturing ‘expert opinion’. A fauna fire-response database was compiled and provided a starting point for allocating species to functional groups. The ‘Bradstock classification’, based on species refugium requirements during a fire and recovery stage requirements after a fire (Bradstock *et al.* 2005), enabled the continuum of fire responses to be categorised into a manageable number of groups. This provided an objective and reliable means to assess the needs of a range of species and identify those likely to be fire sensitive.

Field surveys in burnt and unburnt areas confirmed that bird species predicted to be fire sensitive were strongly associated with (or were restricted to) unburnt spinifex.

Five fire-sensitive bird species were investigated further to identify their fire-patchiness requirements. However, the ideal characteristics for unburnt patches within fire scars could not be defined. The methodology provided insight regarding fire size, nonetheless. To apply the fire sensitive species method, fires scars of at least 100 ha were required. Burnt areas less than 100 ha were too small to survey as birds were called in from the fire scar edge. Restricting fire size to around 100 ha was considered acceptable as an initial goal to conserve birds in the bioregion. At this size, fires would kill some individuals, but were unlikely to have a detrimental affect on the species' population. At smaller fire sizes species could be conserved *in situ.*

As found in other fire-prone areas (e.g. Bradstock *et al*. 2005; Tasker & Baker 2005; Mills 2004), an intermediate fire frequency (5 - 20 years?) should cater for the full range of bird species better that frequent recurrent fire or total fire exclusion. Fire in spinifex/snappy gum landscapes was considered beneficial for rejuvenating habitat, but the optimum fire frequency is unknown.

The approach taken here is an advance because a target fire size can be proposed. Most other authors are less specific and recommend 'small' burns (Russell-Smith 2002; Bradstock *et al*. 2005; Woinarski *et al.* 2005). However, Woinarski *et al.* (2005) did recommend that a species home range be considered (which could be as small as 1 ha) and Fraser *et al.* (2003) recommended 7 ha as a suitable fire size for Partridge Pigeons. This raises the question whether managers should consider species at the population level or as individuals. Average fire size in the bioregion is currently more than three orders of magnitude larger than the 1 km^2 size recommended here, and much larger wildfires occur. I contend that the very small fire sizes that individual animals require are unrealistic for landscape fire management. Focusing management at the population level would allow for species persistence and provide a more attainable objective. Fires of 100 hectares may also appear to be miniscule and unrealistic. But the value for 'average' fire size is skewed by a few large fires. Almost half (45%) of the fire scars proposed for bird survey work in 2005 were less than 100 ha, confirming that many small fires already occur.

6.3.5 Options for Monitoring

Applying the fauna functional group concept enables clear objectives to be set for ecological fire management. Fires of 20 ha or larger can be readily mapped using existing remote sensing technology (Yates & Russell-Smith 2002; http://www.firenorth.org.au) and ecological outcomes could be monitored by recording the presence or absence of fire sensitive species. For example, fauna could be monitored at the property and regional scale using a suite of **RtSt** bird species (those with specific, temporary refugium requirements during a fire, and needing a certain recovery stage post fire) as indicators. The presence of any of these species after each fire season would imply that sufficient unburnt habitat was available. Being able to choose from a group of species with similar post-fire requirements is efficient. Incidental sightings of any of the species could be recorded, with a targeted search effort for particular species if they had not been sighted in three consecutive seasons.

Ooi *et al*. (2006) suggested that fire patchiness be incorporated into fire mapping because spatial variation within a fire footprint can be significant to plant population persistence. However, my study showed that monitoring internal patchiness at a fine scale in the Mount Isa Inlier was not practical. Applying fine scale imagery such as IKONOS may be conceptually promising, but image analysis techniques are yet to be adequately refined for this monitoring purpose. The imagery I used was expensive and required significant computer processing power and expertise to move from the format provided to a useable end product. There is a similar attraction to map fire severity using satellite imagery (Hammill & Bradstock 2006), but this also proved difficult with additional issues to do with synchronisation between the dates of burning and image capture. A number of technical issues need to be resolved to make fine scale monitoring of fire patchiness or fire severity operationally available.

6.4 The Patchiness Paradigm

My findings provide a way to progress from setting management goals based on ecological principles, such as 'a fine scale mosaic', to setting more specific targets based on ecological understanding and a coherent process. The procedure was modelled on Lambeck's (1999) focal species concept, which was intended to guide conservation efforts in fragmented landscapes. Issues faced by Lambeck were similar to those facing fire managers. General ecological principles encouraged 'enhancement planting' of remnant vegetation, but this was non-specific and had uncertain outcomes. Encouraging fire regime heterogeneity and fine-scale landscape patchiness is similar. It provides general direction but no detail, which hinders later assessment. Lambeck proposed 'strategic enhancement' as an alternative. He used expert opinion to pre-empt fauna responses, categorised species to identify the most vulnerable, and then specified the minimum requirements to meet their needs. Lambeck's classification categories proved inappropriate in more natural uncleared conditions, but my application of the 'Bradstock classification' used the same rationale. A 'strategic mosaic' can be defined by identifying fire sensitive (focal) species and their landscape requirements. Thus, fire management can be directed towards ensuring their long-term conservation. Note that the vulnerable bird species in the Mount Isa Inlier were relatively sedentary habitat specialists, similar to Lambeck's findings.

Identifying a strategic mosaic provides better guidance than developing and maintaining fire regime heterogeneity (see Bradstock *et al*. 2005). This latter, all-purpose approach suggests that a range of fire frequencies, intensities and seasons over time will benefit species conservation. There are problems with this for the Mount Isa Inlier. Most burning currently occurs at the onset of summer storms and into the wet season. This may be more suitable timing than burning over a range of seasons, since the vegetation recovers quickly after rain, which is imminent at this time of year. Fires in the early to mid-dry season may be difficult to contain and also expose the soil surface for a number of months. And, as previously noted, planning for variation in fire intensity is difficult in spinifex/snappy gum vegetation. I suggest that fire size should take precedence in this landscape. The patchiness paradigm provides general direction but must be interpreted in the regional context. In the Mount Isa area, an annual patchwork of burns in different locations appears to be more appropriate than maximising fire regime heterogeneity.

The focal species approach to setting fire management objectives assumes that effectively managing vulnerable species will cater for the widest range of additional species. This needs further testing. **RtSt** species have special requirements and are immediately sensitive to fire but this category did not include species that might be affected over longer time periods. However, catering regularly for **RtSt** species is also likely to suit species subject to more insidious fire affects, such as hollow-nesting birds.

A key benefit of the approach is that management requirements are assessed according to current landscape conditions. Other attempts to guide fire management have focused on identifying changes to the landscape and biota over time, and trying to reconstruct the fire history (Fensham 1997a, 1997b; Crowley & Garnett 2000; Vigilante 2001). These authors all recommend fire management aimed at creating a ‘fine-scale mosaic’, based on the philosophy that burning by Aboriginal people resulted in landscape heterogeneity at a much finer scale than currently expressed. The focal species approach leads to the same outcome but emphasises the present condition and progressing from there. The method can be applied at property and landscape levels.

6.5 Improving Fire Management

The research topics for this study were influenced by considering the hierarchy of scales. This concept now provides a convenient framework to integrate the results and reflect on fire management in general. It also enables predictable factors to be distinguished from stochastic factors (Table 6.1), which will assist adaptive management. Predictable factors directly aid in setting management objectives while unpredictable factors need to be accepted and accommodated. For example, lightning strike locations cannot be forecast, but lightning does ignite wildfire. The possibility of such a wildfire needs to be taken into account.

This cross-scale structure may assist adaptive management by explicitly portraying the current state of knowledge. There are limits to our abilities to predict specific behaviours and outcomes (Christensen 1997), so uncertainty is inherent in the system. However, further research could move currently unpredictable factors towards a higher degree of predictability.

Table 6.1 Integrating fire-related information across scales and deciding which factors operate predictably assists in formulating fire plans.

Scale of observation	Temporal and spatial aspects Predictable ◄── Conditionally predictable	── Unpredictable
MACRO e.g. bioregion, broad vegetation type	• Wildfire risk is related to strength of the wet season • A low to moderate fire frequency is beneficial • Short fire return interval, decrease **RtSt** species • Long fire return interval, increase wildfire hazard	• The proportion of landscape at each recovery stage • What configuration of 100 ha fires will contain wildfires? • Degree of climate variation on an annual basis
MESO e.g. property, vegetation community characteristics	• Discontinuous fuel, high level of variability within one fire event • Fires abut areas burnt in the previous 3 years • Series of fires in short succession leads to decline in vegetation structure	• The exact level of internal fire patchiness for any one fire • Ignition points (lightning and people) • The exact fire history or 'invisible mosaic'
MICRO e.g. points or sites, individual species responses	• Most perennial plants flower in 3-5 years post fire • A flush of ephemeral species occurs within 1-2 years post fire depending on rainfall • **RtSt** species need long unburnt spinifex • The vegetation response at individual points in the landscape is related to plant characteristics • Intra-fire heterogeneity will ensure fire tolerant and fire sensitive plant species are present	• The longevity of the soil seed bank • Seedling survival post fire (e.g. depends on follow-up rain, competition and grazing hazard) • Time required for fire sensitive fauna species to recolonise

6.6 Recommendations for Ecological Fire Management

Predicting vegetation development in fire prone ecosystems requires an understanding of the factors controlling landscape patterns (Turner *et al.* 2001). This study found that the main factors controlling fire-induced mosaics in spinifex-dominated low open woodlands of the Mount Isa Inlier were:

- **Recovery of the fuel load since the previous fire**. This depended on wet season rainfall. Rainfall was variable from year to year and in some years there were extensive wildfires. Therefore, the proportion of vegetation at each stage of recovery was not in static equilibrium.
- **The discontinuous nature of hummock grasses as fuel.** This affected internal fire patchiness. As a result, perennial species were often present at a range of ages and size classes. The unburnt patches ensure that fire sensitive species or stages of development will survive.
- **Individual plant characteristics**. At the scale of individual points in the landscape, plant vital attributes (Noble & Slatyer 1981) controlled the vegetation expression after fire (i.e. whether plants were sprouters or obligate seeders).
- **Recurrent fire**. At the broad scale, after soil type and aridity, the vegetation was controlled by recurrent fire. Eucalypts with a single trunk or a few sizeable stems indicated a significant fire-free interval, as did large hummocks of senescing spinifex. Snappy gums with a mallee form indicated more recent, or frequent burning.

6.6.1 What Should Fire Managers Aim For?

Fire size

I propose that fires of around 100 ha (or less) would be acceptable for conserving bird species in the area. This needs testing for other fauna, but provides an initial target. Internal fire patchiness is assured by the nature of the vegetation. It was evident from field inspections that small fires commonly occurred, but these were inadequate for preventing large fires. Wildfires need to be minimised as a matter of critical importance. For this, fire patterns need to be annually reviewed and interpreted in conjunction with rainfall records. Large fires achieve an importance beyond their frequency and have far-reaching impacts compared to many small fires (Gill 2000). The number and configuration of small fires required to modify the characteristics of unplanned fire is not known, however. Too few small fires lead to increased wildfire risk. A few very large fires lead to landscape homogenisation (Lindenmayer & Burgman 2005).

A confounding issue is the potential for increased grazing pressure on regenerating vegetation, with palatable species selectively removed (Partridge 1999). There is a trade-off between having burned patches large enough to maintain plant diversity (by spreading risk for species that are preferentially consumed by herbivores) and small enough to conserve species of fauna. Management of post-fire grazing by domestic species can partially address the issue, but further research would provide insight.

Fire frequency

A sequence of fires in short succession would result in vegetation with a more open structure. The cover of tree and shrub species would eventually decline. Tree seedlings would not be recruited to the canopy and fewer obligate seeders would establish. However it is unlikely that species composition would change. Vegetation recovery time is relatively quick in savanna habitats (Mills 2004) and species can be retained in the soil seed bank. Fuel does not accumulate sufficiently to sustain a fire for at least three years from the previous fire, by which time most species have set seed.

Fire frequency has more immediate effects on fauna. Animals requiring long-unburnt spinifex or tree hollows are predicted to decline if fire intervals are short. Long fire-free intervals would increase the proportion of large tree boles and tree hollows, but also increase the risk of wildfire. In such fire prone vegetation, fire exclusion is unrealistic and a subsequent fire inevitable. I suggest a 10 to 20 year fire return interval as an initial target, erring on the longer timeframe if protection from wildfire is possible.

Fire intensity

Emphasis in other parts of Australia's tropical savannas is to reduce fire intensities by increasing early dry season burning and reducing the number of late dry season fires (Russell-Smith *et al*. 2002). This seems less important in the Mount Isa Inlier. Differentiation in fire intensity between fire scars appeared to be minor. All fire scars had a high degree of internal variability, but this was attributed to discontinuity in the fuel bed and variation in weather conditions during the fire, rather than season of burning.

Low intensity savanna fires were considered to increase the diversity of habitat available to birds because both bare and sheltered ground was available (Mills 2004). This outcome can be achieved by reducing fire size in spinifex-dominated woodlands.

Fire season

The interplay between the seasonality of fire and response of vegetation or fauna was not investigated. As discussed above, I suggest that restricting fire size is the key concern for the study area.

The landscape context

Hansson (1997) suggested that landscapes be the focus of management. This is highly relevant for fauna conservation. Planning small fires needs to be considered with regard to the fire history of the surrounding landscape. At least one edge should adjoin long-unburnt country to facilitate recolonisation by fire-sensitive species. Long unburnt areas might become priority protection areas, depending on the overall age structure of regional vegetation. Up-to-date fire history maps will aid this planning, as will spatial modelling to examine the limits to this requirement in a landscape.

Monitoring outcomes

Monitoring is a core component of adaptive management. As above, recording fire size using satellite imagery and coupling this with ground-based surveys of fire-sensitive species will confirm whether the fire patterns are adequate for species conservation. Very fine scale imagery and measures of internal fire patchiness are not required from an operational perspective, but could provide insight for research, especially if the requirements of endangered species are in question. Satellite derived data should be applied at a scale appropriate to the issue (Ludwig *et al*. 2007). Adopting the 'thresholds of potential concern' approach used in South Africa (Parr & Andersen 2006) has merit, but upper and lower thresholds would need to be determined for the variables or species of interest. The desired landscape condition is a dynamic concept and requires a flexible management system that acknowledges uncertainty. Annual review of fire plans, annual implementation and annual monitoring of outcomes is necessary due to the climatic variability of the area.

6.7 Recommendations for Further Research

Research priorities for fire management in the Mount Isa Inlier can be grouped into three broad areas: how to implement the mosaic, how do species actually respond to fire, and how can information be effectively integrated across scales and disciplines?

6.7.1 Implementing the Mosaic

The distribution of patches in the landscape

My findings suggest a target fire size of around 100 ha in the first instance. Smaller fires are also desirable and fires up to 200 ha are not anticipated to be detrimental to vertebrate species at the meta-population level. Many fires smaller than 100 ha already occur; nearly half of the fire scars initially identified for bird surveys were in this category. However, fire mapping data showed that there were numerous fires 100 and 200 times larger and occasional fires were over 100,000 times larger. What are the within-fire characteristics of these fires? It seems logical from the landscape patchiness paradigm that these very large fires should be prevented, but this requires further research. Will a particular configuration of 100 ha patches limit the spread of wildfire? Should a target number of fires be imposed each year or does this depend on the strength of the wet season (with greater saturation of small fires in years of above average rainfall)? A fire return interval of 20 years suggests that 5% of the landscape could be burnt annually. Mapping records for the six years 1998-2003 showed 1.6% of the bioregion burnt in one year and on average 10% burned per year. These figures engender optimism; however in the largest fire year, 35% of the bioregion burnt. Are years of extensive wildfires inevitable? Computer modelling may assist to answer these questions (Cook & Liedloff 2001, cited in Andersen *et al.* 2003).

Improved techniques for mapping fire scars from remotely sensed data are required for monitoring these smaller fires. An automated process is underway (Peter Thompson, pers. comm. 2006), but field verification is necessary as many small fires are currently overlooked.

Fire behaviour

Limiting the size of fires needs a thorough understanding of fire behaviour. The present pattern of burning late in the year at the time of early summer storms and the onset of the wet season may benefit rapid recovery of burnt vegetation. However, winter nights may be sufficiently cool to extinguish fires, making it feasible to restrict fire size at this time of year. If so, what is the trade-off between winter burns for small fire size and summer storm burns for rapid recovery?

Meteorological studies

Linked with fire behaviour is better knowledge of the weather patterns in the Mount Isa Inlier. Temperature, rainfall, relative humidity and dew point records are available from the Australian Bureau of Meteorology. These data span the past 100 years and could be investigated for a better grasp of weather variability. Some of these data have already been used for modeling climate (e.g. for anticipating rainfall based on the Southern Oscillation Index, Clewett *et al*. 2003) and pasture growth (Carter *et al*. 2003). Such information could also assist with evaluating the effects of impending climate change due to global warming.

Lightning data can now be sensed remotely (Casper 1990; Montandon 1992) and could be matched with fire hot spot data to estimate the proportion of fires ignited by lightning strike. Reports in the literature suggest that less than 1% of fires in the Australian tropical savannas are caused by lightning and the remainder by people (Williams *et al*. 2002). From discussions with landholders in the study area, I surmise that lightning is the cause of approximately 50% of fires in the Mount Isa Inlier. This estimate is similar to that for the arid zone in south-eastern Western Australia (Haydon *et al.* 2000). Knowing the probability of lightning caused fires would assist in developing fire management strategies. If very few lightning ignitions occurred, public education and strategic fuel reduction would be required to minimise the incidence of wildfire. High rates of ignition by lightning would necessitate that more attention be given to managing for uncertainty.

Operational adaptive management

Detailed knowledge of the fire mosaic requirements for species conservation is merely the first step in implementing ecological fire management practices. The application of science to management needs to consider the local and regional, social and cultural settings (see Smith *et al*., Appendix 1). The success of adaptive management depends on individuals and organisations being able to develop and implement plans, monitor outcomes in relation to the initial objectives, review success or otherwise, and act on the results. Systems analysis research is required to understand the multiple facets of applied fire management and help move research findings to social reality (Smith *et al.* in press). Different tenure types, land use objectives and landholder attitudes further confound the capacity to implement ecological fire management at the requisite bioregional level. Fire management in the tropical savannas is an annual, iterative process so this research is important for long-term species conservation.

Social and economic research

Again, fires operate at the landscape scale and people are an integral part of the landscape. Their values and aspirations will influence if, when and how fire management is undertaken. Influencing attitudes towards fire management for ecological outcomes requires both social and economic research (http://www.landmanager.org.au). What is the prevailing attitude towards fire? What are the real costs of establishing and maintaining the desired mosaic, including training needs? What are the benefits? Based on cost-benefit analyses, does ecological fire management also benefit cattle production? Is it the most viable fire management alternative?

Aiming for small scale fires is in keeping with the fine scale of landscape patchiness resulting from traditional Aboriginal land management, which is still practiced in some areas (Russell-Smith 2002). Social acceptance of Aboriginal occupancy prior to Australia's settlement by Europeans has resulted in specific legislation to recognise, protect and conserve Aboriginal cultural heritage (e.g. Queensland's *Aboriginal Cultural Heritage Act 2003*). Depending on the perspectives of indigenous people in the region, there is potential to involve them in fire management across different land tenures. Social research is required to record traditional knowledge for the area and obtain the views of the Traditional Owners regarding future management. Then, different models for integrating this knowledge with other land management objectives, and involving Aboriginal people in day to day fire management practices, need to be formulated and assessed. A positive outcome could address the dual goals of maintaining biodiversity and conserving cultural heritage.

6.7.2 Species' Fire Responses

Chapter 3 considered the fire responses of the dominant plant species and Chapter 5 explored the requirements of key fauna. A broad approach was used in both due to the lack of published information. There were innumerable 'unknowns'; therefore there are ample opportunities for future research directed towards particular species. As a priority, the recommended 100 ha fire size should be tested to determine if it is adequate for other species. If auto-ecological studies are undertaken, I suggest they focus on fire-sensitive species in order to refine understanding of their requirements. These species are good indicators of an appropriate fire regime. However, further work is required if they are used to monitor the outcomes of fire management. For example, if the concept of 'thresholds of potential concern' is applied, should the response variable be population size, rate of population growth or decline, or species presence or absence in the landscape (Parr & Andersen 2006)? At what levels should these be set to trigger changes to fire management practices?

6.7.3 Cross-Scale, Cross-Discipline Studies

My final recommendation relates to an approach to research rather than a specific research topic. This study used an hierarchical framework to integrate data collected at different scales and across different subject areas. This proved useful for gaining an overview of fire management in a region where prior research was scant. A major advantage of such a framework is that it assists in project planning (key elements of enquiry can be readily identified) as well as translating research findings for applied management. The cross-scale, cross-discipline approach is recommended for rapidly assembling information and providing a general understanding of how an ecosystem is operating.

6.8 Conclusion

This study sets the scene for putting ecological fire management into practice in the Mount Isa Inlier. It builds on the current body of knowledge for fire management in Australia's tropical savannas and, importantly, shows that bioregional differences need to be taken into consideration. The body of research for the mesic northern savannas and arid interior of Australia does not directly apply to the semi-arid Mount Isa Inlier. The nuances are important for on-ground implementation as well as species conservation.

The major contribution of the work is that it bridges the gap between ecological principles (the patchiness paradigm) and practical land management. It provides an objective method for identifying fire sensitive species and then recommends that management be tailored to their needs. It is one of the few studies that endeavours to quantify the level of patchiness to which managers might aspire. The study demonstrates a strategic approach that meshes well with adaptive management and can be applied in different environments and at different scales. Finally, it illustrates the benefit of integrating data collected across different scales and subject areas for rapidly informing fire management.

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APPENDIX 1

Papers Produced During the Course of This Work

Paper 1

C. S. Smith, L. Felderhof & O. J. H. Bosch, 2007. Adaptive Management: Making It Happen Through Participatory Systems Analysis.

Accepted in Jan 2007 for publication in 'Systems Research and Behavioral Science' (Syst. Res. **24**, 1-21, 2007). I was responsible for approximately 50% of this work.

Paper 2

L. Felderhof & D. Gillieson (2006) Comparison of fire patterns and fire frequency in two tropical savanna bioregions *Austral Ecology* (2006) **31**, 736–746

This paper is based on the findings in Chapter 2.

Paper 1

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Paper 2

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APPENDIX 2 Species Longevity and Fire Response

a = annual, e = ephemeral, p = perennial; o = obligate seeder, r = resprouter; u = unknown

GROUND LAYER SPECIES	Longevity	Fire response		Longevity	Fire response		Longevity	Fire response
Alternanthera nodiflora	a	o	*Fimbristylus sp.2*	u	u	*Solanum quadriloculatum*	p	o
Aristida sp.	p	r	*Galactia sp.1*	a	o	*Sorghum sp.1*	u	u
Aristida sp.1	p	r	*Glycine sp.*	p	r	*Spermacoce sp.*	a	o
Aristida sp.2	p	r	*Goodenia armitiana*	a	o	*Sporobolus australasicus*	a	o
Aristida sp.3	p	r	*Goodeniaceae sp.1*	u	u	*Sporobolus sp.*	u	u
Astrebla sp.1	p	r	hard spinifex	p	r	*Tephrosia sp.2*	p	r
Astrebla sp.2	p	r	*Heteropogon contortus*	p	r	*Themeda australis*	p	r
Blumea sp.1	a	o	*Indigofera colutea*	e	u	*Themeda sp.*	p	r
Boerhavia paludosa	p	u	*Indigofera linifolia*	e	o	*Tribulopsis sp.1*	p	u
Boerhavia pubescens	u	u	*Indigofera sp.*	e	u	*Trichodesma zeylanicum*	p	o
Bulbostylis sp.1	a	o	*Indigofera sp.1*	e	u	*Triodia pungens*	p	r
Cenchrus ciliaris	p	r	*Ipomoea sp.*	e	u	*Triodia sp.*	p	r
Chamaesyce sp.	a	u	*Ipomoea sp.1*	e	u	*Zornia muriculata*	p	o
Chrysopogon fallax	p	r	*Ischaemum australis (?)*	p	r	*Zornia sp.1*	p	u
Crotalaria sp.	p	r	*Iseilema sp.1*	a	o			
Cymbopogon bombycinus	p	r	*Melhania sp.1*	p	o			
Cyperus sp.	u	u	*Mnesithea formosa*	a	o			
Dichanthium sp.	u	u	*Mukia maderaspatana*	p	r			
Dichanthium sp.1	u	u	*Neptunia sp.1*	p	r			
Dichanthium sp.2 (?)	u	u	Other forb' sp.1	u	o			
Dichanthium sp.3	u	u	Other grass'	u	r			
Digitaria sp.	u	u	*Panicum decompositum*	p	u			

GROUND LAYER SPECIES	Longevity	Fire response		Longevity	Fire response		Longevity	Fire response
Digitaria sp.1	u	u	*Paspalidium sp.1*	u	u			
Dodonaea lanceolata (?)	p	o	*Phyllanthus maderaspatens*	a	o			
Enneapogon polyphyllus	p	r	*Portulaca oleracea*	a	o			
Enneapogon robustissimus	p	r	*Rhynchosia minima*	p	u			
Enneapogon sp.1	p	r	*Scaevola sp.1*	a	r			
Eriachne ciliata	a	o	*Schizachyrium fragile*	a	o			
Eriachne sp.1	u	u	*Setaria surgens*	a	o			
Eriachne sp.3	u	u	*Sida sp.*	p	u			
Fabaceae sp.1	u	u	*Sida sp.1*	p	u			
Fimbristylus sp.	u	u	*Sida sp.4*	p	u			
Fimbristylus sp.1	u	u	*Solanum (echinolatum ?)*	a	o			

SHRUB LAYER SPECIES	Longevity	Fire response	TREE LAYER SPECIES	Longevity	Fire response
Acacia chippendalei	p	o	*Acacia shirleyi*	p	o
Acacia chisholmii	p	o	*Atalaya hemiglauca*	p	r
Acacia elachantha	p	r	*Corymbia bella*	p	r
Acacia galioides var galioides	p	o	*Corymbia sp.1*	p	r
Acacia gonoclada	p	o	*Corymbia terminalis*	p	r
Acacia hammondii	p	o	*Eucalyptus (grandifolia ?)*	p	r
Acacia hemignosta	p	r	*Eucalyptus aspera*	p	r
Acacia hilliana	p	o	*Eucalyptus dichromophloia*	p	r
Acacia holosericea	p	o	*Eucalyptus grandifolia*	p	r
Acacia monticola	p	o	*Eucalyptus leucophloia*	p	r
Acacia phlebocarpa	p	o	*Eucalyptus leucophylla*	p	r
Acacia retivenea	p	o	*Eucalyptus pruinosa*	p	r
Acacia sp.	p	o	*Eucalyptus sp.*	p	r
Acacia tenuissima	p	o	*Eucalyptus terminalis*	p	r
Acacia thomsonii (?)	p	o	*Grevillea striata*	p	r
Carissa lanceolata	p	r	*Hakea arborescens*	p	r
Cleome viscosa	a	o	*Hakea sp.*	p	r
Corchorus sp.	e	r	*Hakea sp.1*	p	r
Crotolaria novae-hollandiae	a	o	*Lophostemon grandiflorus*	p	r
Flueggea virosa	p	r	*Lysiphyllum cunninghamii*	p	r
Grevillea dryandri	p	o	*Terminalia aridicola*	p	r
Hakea arborescens	p	r	*Terminalia canescens*	p	r
Heliotropium sp.	p	r	*Terminalia sp.*	p	r
Indigofera pratensis	p	o	*Ventilago viminalis*	p	r
Indigofera sp.	e	o			
Jacksonia vernicosa	p	o			
Lysiphyllum cunninghamii	p	r			
Maytenus cunninghamii	p	r			
Melaleuca viridiflora	p	r			
Petalostigma quadriloculare	p	r			
Scaevola ovalifolia	p	o			
Senna sp.	p	u			

APPENDIX 3

Cosine Correction – An Image Pre-Processing Step to Reduce the Effects of Shadowing Due to Differential Terrain Illumination

Equation (from Law & Nicol 2005):

$$\mathbf{Cos\ i = cos\ E\ cos\ Z + sin\ E\ sin\ Z\ cos\ (A_o – A_s)}$$

$$\mathbf{Ln\ (\lambda) = L\ (\lambda)\ /\ cos\ i}$$

$$\mathbf{L_H = L_T\ cos\ z\ /\ cos\ i}$$

Where

L = radiance

Z = solar zenith angle

L_H = radiance for horizontal surface

L_T = radiance observed over the inclined terrain

i = incidence angle with respect to surface normal

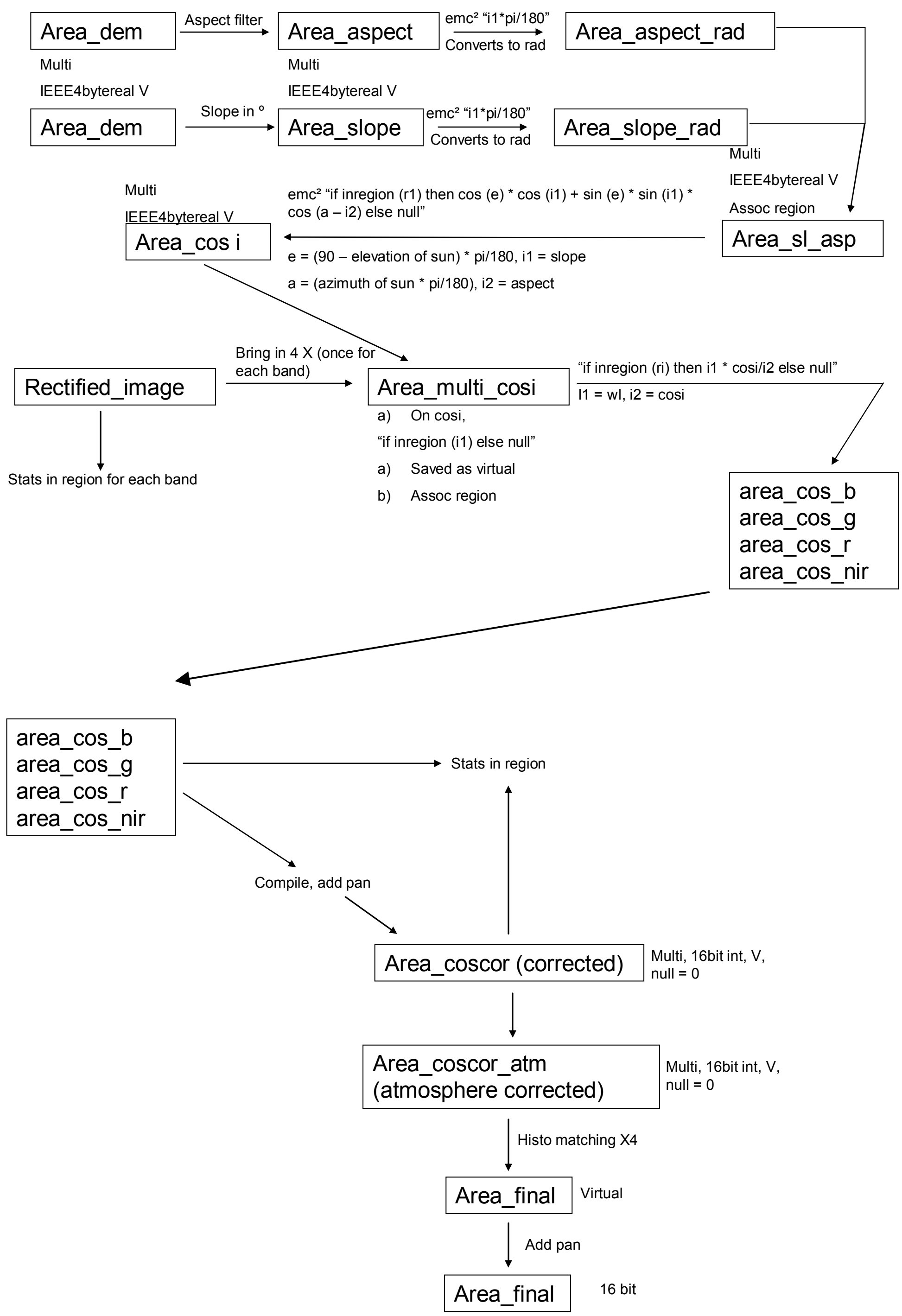
E = slope inclination

Z = solar zenith angle

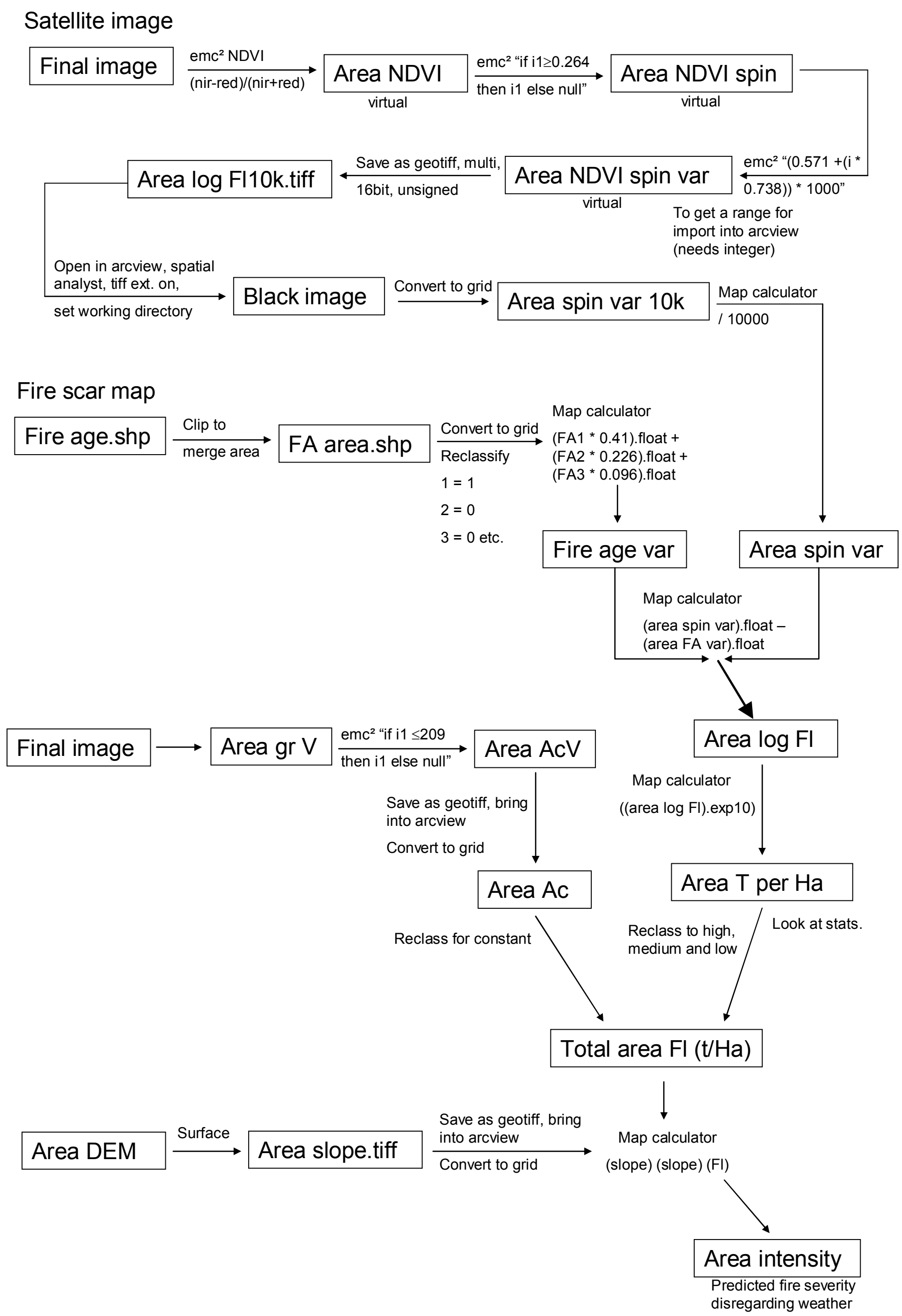
A_o = solar azimuth

A_s = surface aspect of the slope angle

Flow diagram showing generic file names (in boxes) and processing steps undertaken to apply the cosine correction method to the IKONOS imagery:



The processing steps required to produce the fuel load map based on the regression equation for spinifex:



APPENDIX 4

Fauna Database

See enclosed CD