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Proceedings of an Australian Bushfire Conference, Perth, Western Australia
27-29 September 1993

EDITED BY W. L. McCAW, N. D. BURROWS,
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Landscape Fires '93: Proceedings of an Australian Bushfire Conference, Perth, Western Australia 27-29 September 1993

EDITED BY W. L. McCAW[1], N. D. BURROWS[2], G. R. FRIEND[3] AND A. M. GILL[4]

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PREFACE

The Landscape Fires '93 Conference was the fourth in a series of Australian bushfire conferences which commenced in 1987 at the Australian Defence Force Academy campus of the University of New South Wales in Canberra. The 1993 conference in Perth was the first to be staged outside Canberra and was sponsored by the Western Australian Department of Conservation and Land Management (CALM) and the Bush Fires Board of Western Australia. More than one hundred delegates from Australia, New Zealand and the United States participated in the conference sessions and the subsequent field trip to the Mount Lesueur National Park some 300 km north of Perth. A list of conference delegates together with their affiliations has been included in these proceedings.

The purpose of the conference was to provide a forum for members of the Australian fire community, both researchers and managers, to review the latest concepts and developments in bushfire science. Invited speakers presented papers addressing six main themes identified for the conference:

- fire danger rating and fire behaviour prediction
- fire measurements for fire ecology studies
- bushfires and the urban-rural interface
- bushfire emissions
- fire-plant-animal interactions
- fire-induced landscape mosaics

A large number of poster papers were also displayed in conjunction with each session.

These proceedings contain the text of most papers presented at the conference. A few speakers elected not to publish the full text of their papers because the work was not yet ready for publication, or because time constraints prevented preparation of a final manuscript. All papers published in these proceedings were reviewed by two independent referees, where possible at least one of them having been a delegate at the conference. Abstracts of poster papers, as presented at the conference, have been included in these proceedings .

The organizers of Landscape Fires '93 would like to sincerely thank the large number of people from many organizations who contributed to the success of the conference. Those people who kindly assisted the editors by reviewing papers are acknowledged in a list at the end of this volume. Particular thanks are extended to Joanne Elliot and Michelle Lathwell from CALM's Science and Information Division at Manjimup for their tireless work which included registering delegates, making arrangements for the conference, and preparing these proceedings for publication submission.

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Opening remarks

R. J. UNDERWOOD[1]

[1] General Manager, Department of Conservation and Land Management, Crawley 6009, Western Australia. (now retired)

Thank you for the invitation to open this conference. It is a pleasure to be involved. I have had a passionate interest and a personal involvement in fire management for over 30 years, and my participation has spanned the full spectrum from fire-fighter, through district management and research to policy and politics. Thinking back on this, I can't quite decide which role was the most stressful - but I do remember working with many tough, dedicated and thoughtful people. It has always seemed to me that 'fire people' are a very special subculture of society, and are very special people; not many people choose a tough assignment with few rewards - and that is what fire work is mainly about.

I would like to open by acknowledging the history of these conferences. They were initiated six years ago by the Mathematics Department of the Australian Defence Force Academy in Canberra, of all places ...although you don't need to think about it for very long to see the connections. I understand that the concept originally was to provide a forum in which people working in fire science could get together, but it was quickly enlarged to include fire managers, a move which makes a lot of sense. After all, fire scientists and fire managers are interdependent organisms in the same ecosystem. This is now the fourth Landscape Fires Conference and the first to be held off the Defence Academy campus. I am delighted that this conference is being hosted by CALM and the WA Bush Fires Board (BFB).

Speaking on behalf of CALM, I would particularly like to welcome visitors from other parts of Australia, the USA, and New Zealand to WA. We are very proud of our traditions and of our record in fire management and fire research in WA. However, this is not to say that we are resting on our laurels. In fire science and fire protection there is still a long way to travel and there are many complexities yet to be tackled, and resolved. We have a huge State, a hot, dry season every year and the usual intermixture of people values and ecological values. We are also chronically short of money to do the sort of research and fire management programs that we think should be done. However, we do have some advantages. For example, CALM is an integrated agency and can undertake fire management and planning across various tenures, and we do have a well integrated multi-disciplinary research group. There are excellent relationships between colleagues in the Bush Fires Board, local government and the volunteer fire-fighters. And compared to many government bureaucracies, CALM staff are accustomed to and committed to change. We also like to share our problems with our peers and our friends in the wider fire community, and to hear their views on what we should or should not be doing, or what we should or should not be thinking about when it comes to fire. So please don't hold back on your helpful advice!

Turning now to the general issue of fire, I have to say that it is one of the most difficult and persistently controversial issues that CALM faces. The other two, incidentally, are timber cutting in native forests and providing for tourism and recreation in conservation reserves. From a managers perspective all three issues are similar: in each case, CALM has the job of working out a path to follow on the ground, while negotiating a minefield of complex interactions between social values, legislation, environmental values, science and conflicting community expectations. More often than not we find ourselves caught in the crossfire between interest groups, or trying to develop compromises with uncompromising people.

However, although the three issues are similar in a political and management sense, fire is the most demanding. This is because the cost of getting it wrong is so large, and because accountability for getting it wrong is so obvious! If you get your silviculture wrong, or your recreation planning is below standard, it's not usually a matter of life or death. But if people or townships are consumed in a bush fire, and CALM's or BFB's policies or fire protection measures are found to have been deficient or irresponsible, we are in huge trouble. And by 'we' I don't just mean the Department or the Board, I mean me, and the foresters, rangers and brigade officers out there in the parks and districts who personally carry the heavy burden of responsibility. Furthermore, thanks to the regrettable Americanization of the approach to litigation in Australia, getting it wrong can also mean that the government treasuries are in trouble. Bushfires sometimes cost lives, but they nearly always cost dollars; serious bushfires can cost millions of dollars. The only people to profit from this are the lawyers; everyone else loses.

For these reasons land managers with fire responsibilities tend to be a fairly conservative breed - they like to stick to what they know works best. This can hamper relationships between managers and scientists, and between managers and environmentalists and it can cause big problems when value judgements have to be made, for example, air quality in Perth versus bushfire mitigation in south-west forests. Both are desirable, but it's very hard to advantage one, without disadvantaging the other. It is not that ecological and environmental values are not understood or held important by managers; it's a matter of keeping the community's back covered - and your own at the same time.

I would like to conclude my remarks with comments on two important issues related to fire management and fire science. I am sure these issues will be discussed during the conference.

1. With respect to managing fire, I believe that we must tackle two priority issues: firstly, we need to develop systems in which fire is truly integrated with all other land use values; and secondly, we need to develop mechanisms to ensure that all the stake holders are actively involved in the decision-making and the implementation processes. For both these reasons I am a very strong advocate of using structured Wildfire Threat Analysis as a basis for fire management planning, and of taking these processes into the community. In this way the priorities, trade-offs, and the special concerns become transparent. In this way we will steadily gather support from interest groups and local communities. Only in this way will we succeed in getting people to understand that fire is not just CALM's or the BFB's problem, it is everybody's problem.

   Having said this, I accept that the greatest difficulty we have in implementing our fire policies comes from urban people who are completely unthreatened by bushfires, who do not understand fire, and who never see it as their problem. Many of these people have great political influence and are beloved of the media. I do not yet have the solution for this problem. But I do know that if rural communities (including rural environmentalists) can work together cooperatively, the fire protection and environmental outcomes will always be higher than if they don't.

   I am also aware that however analytical we are, and however hard we work to ensure community ownership of a fire management plan, some people or groups will stay outside the process and will fight the final plan. This is annoying, but it's democracy, so you have to grin and bear it. I think the problem will decline when the nonsensical idea held by some people that fire is a foreign element to Australian ecosystems also declines. Also (we all need to remind ourselves) no plan can ever be considered a 'final plan' - there will always be new research findings to incorporate, values will change and wildfires will always occur on bad days and cut across the best-laid plans.
   I am very pleased to see a special section of this conference looking at bushfires and the urban interface. This is where everyone has the most to lose and probably where fire managers and scientists have the most work to do.

2. Turning now to fire science, I also would like to make two points. Firstly, it is my view that fire behaviour research is of critical importance and must be continued. Fire is a natural feature of nearly every Australian ecosystem, and because of this, and because of the fact that we now have a fire-vulnerable human society interspersed through natural ecosystems, we will always be wanting to use fires. Whether this use is for ecological purposes, or for wildfire mitigation, it doesn't matter; if we go out there lighting fires we must be able to predict how they will behave and what will happen to the smoke. We must know whether we are asking our own people to do the impossible when we put them up against a fire. We are quite well advanced in this regard for a few ecosystems, but for most we are still in 'elementary school'.

   Secondly, I believe that the time has come for fire ecologists to move beyond studies which focus on individual species and the impact of single fires. These studies are taking us into a morass of increasing complexity and do not help us to design practical and sensible fire regimes. There are an almost infinite number of permutations and combinations between fire and the various elements of the ecosystem and the environment. Post-fire response and recovery can go down many different paths, depending on the intensity of the fire, the frequency of the fire, the post-fire weather conditions and interactions with other factors such as feral animals or insect plagues. This is all of enormous intellectual interest, but doesn't help the manager out there trying to develop an action plan for next summer, or community groups working on a fire plan for a major conservation reserve.

   My view is that the most helpful approach which researchers can take is one which recognizes that fire disturbance is natural in Australian ecosystems and then focuses on the ecosystems and their disturbance boundaries, i.e. identifying the points beyond which ecosystems cannot recover, rather than looking at single species and their response to a single fire event. To this I need to add the need for special research directed at protecting important threatened species - although I caution that the adoption of a fire regime aimed at favouring any one species will inevitably disfavour some other species. As recently as last week I was considering the plight of one of Australia's rarest animals (the Yellow Bellied Frog) which, it turns out, is probably disfavoured by summer or autumn burns - the very sort of fire which we are being urged to carry out by biologists from a different scientific lobby group.

At this point I also need to acknowledge the special problem of small bushland reserves isolated within an agriculture landscape. Here, however, the immediate problem is not fire management, it's rebuilding the landscape within which the reserves occur so that they are better connected and thus more resilient to disturbance.

I love to discuss these issues, and could go on for hours - but I know all of you are dying to get on with your conference. So can I make one final point. I would like to see fire scientists and fire managers work much closer together. There are many opportunities being lost for refinements to techniques, on the one hand, or for field-level experimental management on the other. Even in a close-knit organization like CALM, I see too many examples of researchers and managers pulling against each other, rather than working together. I regard the scientists as our motivators for change while the managers are implementers of change. Successful change will not be achieved unless it is managed properly, that is, presented in a positive and cooperative climate so that it is rapidly incorporated into the daily business of ecosystem management and community protection. I therefore doubly offer a warm welcome to you all at this conference, especially the scientists and managers in attendance, and wish you a stimulating and enjoyable week.

Thank you.

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Separating fire spread prediction and fire danger rating

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ABSTRACT

Australian fire danger rating systems have been developed by relating fire danger directly to predictions of fire-spread. In the past when new fire spread models were developed which changed the basis for predicting fire spread they also changed the basis for calculating fire danger.

This paper examines the consequences of linking fire danger directly to rate of spread by comparing a proposed new fire spread model with the models developed by A. G. McArthur and used in his grassland fire danger rating systems.

The paper concludes that grassland fire danger rating should be separated from predictions of rate of spread. The McArthur MkIV Grassland Fire Danger Meter provides a sound system for calculating fire danger and issuing public warnings on a regional basis.

New fire spread equations should be used to predict fire behaviour for specific fuel types within a local area. They will reflect changes in rate of spread in different pasture types but local arrangements need to be made for appropriate fire restrictions and suppression arrangements when pastures are very discontinuous and will not carry fire.

INTRODUCTION

The concept of **fire danger** involves both tangible and intangible factors, physical processes and haphazard events. By one definition:

> 'fire danger is the resultant of both constant and variable fire danger factors affecting the inception, spread and difficulty of control of fires and the damage they cause.' (Chandler *et al.* 1983)

'Constant factors' are those that change slowly and vary with location; e.g. slope and fuel. 'Variable factors' change rapidly with time but can influence extensive areas; e.g. wind speed and temperature.

The potential for ignition, spread and damage must be present. If there is absolutely no chance of ignition there is no fire danger. If fuels are absent or cannot burn there is no fire danger. If fires can start and spread but there is no economic or other value at risk, as is generally perceived in remote and sparsely populated areas such as some of the open woodlands of northern Australia, there is also no fire danger.

The factors contributing to fire danger may involve chance, e.g. human ignition; be difficult to quantify numerically, e.g. suppression capability; or be intangible, e.g. values at risk. It seems impossible to embody the total concept of fire danger into a single quantitative index.

> **Fire danger rating** is defined as:
> 'A fire management system that integrates the facets of selected fire danger factors into one or more qualitative or numerical indices of current protection needs.' (Chandler *et al.* 1983)

Fire danger rating systems vary in complexity and reflect both the severity of the fire weather and the requirement of management to have some relatively simple measure of the flammability of fuels from day to day. The simplest systems use only temperature and relative humidity to provide an index of the potential for fire ignitions. The most complex systems use better theoretical and empirical models to combine a large number of factors into indices of fire occurrence and fire behaviour. The US Forest Service fire danger rating system, for example, offers the user a choice of six indices or components which are combined into an index of fire load (Deeming *et al.* 1977).

If we accept the above definition of fire danger rating, we then need to examine whether the current systems have selected appropriate factors to meet existing protection needs and to evaluate their performance.

AUSTRALIAN FIRE DANGER RATING SYSTEMS

The McArthur Fire Danger Rating system for dry sclerophyll eucalypt forests was presented to the first Australian Fire Weather Conference in 1957 (McArthur 1958). Fire danger rating tables for grasslands followed soon after (McArthur 1960). These systems were modified and published as both linear and circular slide rules (McArthur 1966, 1967 and 1977) with some modifications to incorporate revised predictions of fire-spread.

Most fire authorities use the fire danger rating system to:

- determine the type and density of fire detection observations;
- determine levels of preparedness for suppression operations within fire districts. This includes setting stand-by rosters for personnel and equipment, hours of duty, pre-location of equipment, etc;
- issue public warnings in an attempt to restrict ignition and, thereby, enhance public safety. This includes restrictions on lighting fires for management or recreation, restrictions on forest and industrial operations, and restrictions on public access;
- provide an appropriate scale for management, research and the law for fire related matters.

Mostly the fire danger rating systems are used for setting levels of preparedness and issuing of public warnings. A survey of rural fire authorities in 1990 found that, by and large, the McArthur fire danger rating systems were satisfactory for determining the functions above (Cheney *et al.* 1990).

The McArthur Grassland Fire Danger Rating Systems (MkIV and MkV) were originally designed for annual grasslands in south-eastern Australia and computed an index between 1 and 100 from fine fuel moisture content and wind speed. Fuel moisture was computed from variables of ambient air temperature, relative humidity and the state of grass curing (Cheney 1991). The grassland fire danger index (GFDI) was directly related to rate of spread (ROS) and the McArthur fire danger meters could be used to predict ROS. The relationships used to predict ROS are given in Nobel *et al.* (1980), and a multiplier of 7.692 is applied to ROS to produce the GFDI on both meters.

Fire danger rating classes were McArthur's expert assessment of degree of suppression difficulty in average pastures (see Table 1).

The MkIV and the MkV meters use different power functions to relate ROS to wind speed and the two meters compute different rates of spread for the same weather conditions. The Mk V meter introduced fuel load (W) - directly proportional to ROS. Both meters have the same FDI which defines the fire danger classes, but these values represented quite different weather conditions. For example, Figure 1 illustrates the different values of the FDI from the MkIV and MkV meters rate of spread predictions for different wind speeds at a fuel moisture of 3.5 per cent.

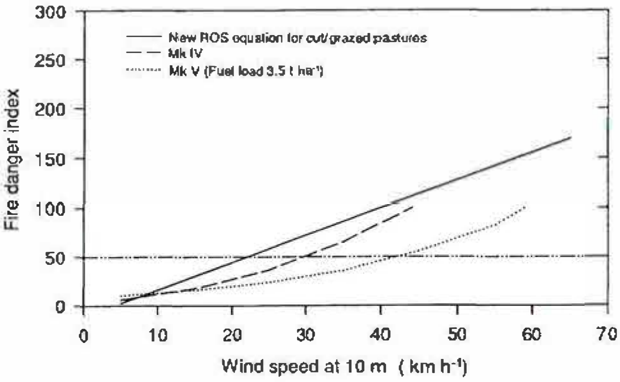

Figure 1. Comparison of the fire danger index for different wind speeds at fuel moisture of 3.5 per cent from the three predicted ROS models (Curing = 100 per cent).

TABLE 1

Grassland fire danger classes, rate of spread (ROS) and difficulty of suppression in annual and perennial pastures that carry a continuous fuel and occur on level to undulating ground (after McArthur 1966).

FIRE DANGER CLASS	FIRE DANGER INDEX	ROS AT MAX FDI IN CLASS ($km\ h^{-1}$)	DIFFICULTY OF SUPPRESSION
Low	0 - 2.5	0.3	Low : Headfire stopped by roads and tracks.
Moderate	3 - 7.5	1.0	Moderate : Headfire easily attacked with water.
High	8 - 20	2.6	High : Head attack generally successful with water.
Very High	20.5 - 50	6.4	Very High : Head attack may fail except under favourable circumstances and back burning close to the head may be necessary.
Extreme	50.5 - 100	12.8	Direct attack will generally fail - backburns difficult to hold because of blown embers. Flanks must be held at all costs.

Effect of New Rate-of-Spread Equations on Grassland Fire Danger

An exhaustive study into grass fire behaviour (Cheney *et al.* 1993) showed that rate of spread was:

- independent of fuel load if grass cover was continuous;
- directly proportional to wind speed when the wind speed exceeded 5 km h^{-1} at 10 m height above ground; and,
- dependent on whether grasses were undisturbed or were close cropped either by cutting or grazing.

These findings were used to develop new equations to predict fire spread (Cheney *et al.* unpublished). Although these equations are still under development they provide a useful example of the consequences of keeping FDI tied to ROS if the new equations are used to predict ROS. The research results and the fire spread model developed from them show that the previous fire danger meters under-estimated spread rates at moderate wind speeds and over-estimated spread rates if extrapolated to very high wind speeds.

Applying the original multiplier of 7.692 on the ROS predicted by the new equations to calculate the GFDI for given combinations of weather conditions (Figs 1 and 2), causes the ratings to be very different from McArthur's original definition (Table 1). The new equations will produce more GFDIs in the Very High and Extreme fire danger classes and as a consequence more total fire bans will be declared.

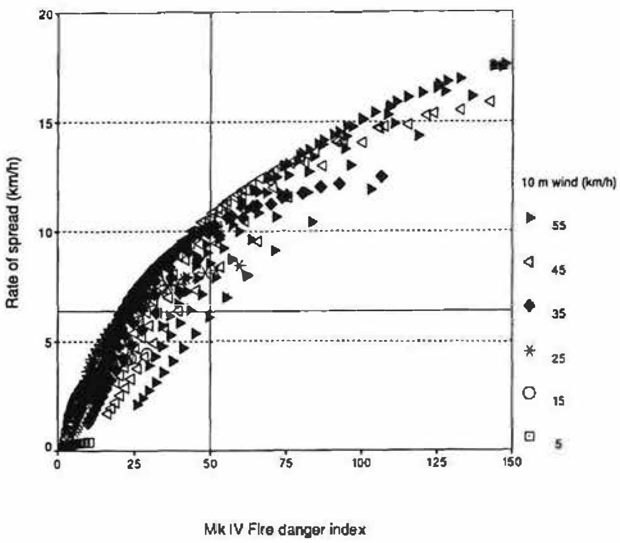

Figure 2. The predicted spread rates by the new equation for cut or grazed pastures for a range of temperature, relative humidity and wind speed conditions related to the MkIV grassland fire danger index for the same weather conditions (Curing = 100 per cent).

The spread rates predicted by the new equations cannot simply be related to the previous GFDI as is shown in Figure 2. The new wind speed algorithm for cut/grazed pastures alone results in a rate of spread of between 6 and 11 km h^{-1} for the combination of weather conditions possible at a MkIV GFDI of 50.

Conversely, if we accept that a rate of spread of 6.4 km h^{-1} represents the boundary between very high suppression difficulty and extreme suppression difficulty as per Table 1 then the new equations will predict extreme fire danger over the range of weather conditions represented by MkIV GFDI between 20 and 50. In general terms this would cause an extreme rating when wind speeds exceeded 30 km h^{-1} over a wide range of temperature and humidity.

Fire authorities were generally satisfied with the performance of the McArthur MkIV meter as a fire weather index (Cheney *et al.* 1990) and this suggests that the McArthur's fire danger classes fairly represented the difficulty of suppression under the prevailing fire weather conditions. Even though we now find that fires have the potential to spread faster than was defined by the original GFDI we consider this does not justify changing the index scale and the weather conditions it represents. Therefore, we believe that fire authorities should retain the GFDI and fire danger classes as defined by the McArthur MkIV Grassland Fire Danger Meter as an index of the severity of fire weather and calculate ROS separately for specific pasture types.

DISCUSSION

Generally, a fire danger rating should be simple and easy to apply (Johnston 1991). We believe that this can only be achieved by separating the functions of public warnings of fire danger from prediction of fire-spread for specific fuel types. Fire control authorities can and do make adjustments for setting general levels of preparedness to account for local variations in fuel load, suppression capability, and resources at risk.

There is considerable misunderstanding of fire danger rating systems. In South Australia, where fire danger rating was given great publicity by news media and roadside signs, few people knew what a fire danger class meant (Dawson 1991). In most cases the only fire danger rating that affects the livelihood of most urban-based people is the EXTREME rating when a Total Fire Ban may prevent the rural barbecue. On the other hand, rural people generally have a good understanding of the preparedness required for different levels of fire danger. This level of preparedness may change during the fire season as the immediate threat to their livelihood changes. For example, wheat growers may request restrictions on the lighting of fires at lower fire danger levels early in the season before they have harvested their crops, than later in the season when they may be keen to burn stubble.

Most public criticism of a fire danger rating system occurs when a Fire Ban District (over which a fire ban is to apply) is too large and there are large differences in the fuel state or weather within the district (e.g. Southern Tablelands of NSW) or when predicted extreme fire danger weather does not eventuate. For example, when frontal systems with dangerous fire weather ahead of a cool change progress more rapidly than forecast, there can be the embarrassment of a warning of extreme fire danger and restrictions on lighting of fires when rain is falling over extensive areas.

Perhaps more confusion arises when fire ban districts are too small (Dawson 1991) and too many fire weather warnings or fire ban restrictions are issued. Many of these issues can be rectified by common sense, with media explanations, or by withdrawing total bans when expected weather conditions do not eventuate.

Forecasts of fire danger indices cannot be very precise because the weather factors contributing to fire danger are additive. Using the Grassland Fire Danger Meter MkIV, errors in forecast variables of:

Curing	100	
Temperature	35	± 4 °C
Dew Point Temperature	15	± 4 °C (RH @ 30 per cent, + 19 per cent or - 11 per cent)
Wind Speed	35	± 10 km h^{-1}

means that the forecast grassland fire danger index (GFDI) of 40 could range from 12 (High) to 90 (Extreme). Considering these errors, one should expect GFDI forecast to vary by at least one fire danger rating class. By and large forecasters do much better than this but even so, one can understand that forecasters and fire authorities may be conservative when the forecast index is within a few points of 50 or whatever index value has been selected for setting a Total Fire Ban.

The same difficulty will, of course, apply to early forecasts of ROS. Predictions of ROS will be most accurate when based on short-term forecasts confirmed or adjusted by measurements of the local weather at the fire site.

The McArthur MkIV Fire Danger Rating System was developed using data mainly from wildfires with some results from experimental fires (Luke and McArthur 1978). The new grassland fire spread equations are designed to predict the potential ROS of grassfires burning in continuous fuels. We believe that the McArthur MkIV Grassland Fire Danger Meter is the most appropriate system for rating the suppression difficulty of grassfires spreading across the landscape in a standard fuel type most commonly represented in that landscape. Some variables have a functional relationship with fire danger which is different to that used for predicting ROS. This is appropriate because some factors will influence suppression difficulty differently than they influence ROS.

The following factors have to be considered when comparing the fire danger index with fire spread models.

Fuel load The fuel factor which most effects fire spread is fuel continuity and not fuel load. Fires will not spread in discontinuous fuels until wind speeds exceed a threshold value related to the degree of discontinuity (Griffin and Allan 1984; Burrows *et al.* 1991). Fuel load is not necessarily related to fuel continuity; for example, a light cover of ephemeral grasses in the inter-tussock spaces may form a continuous fuel bed without significantly adding to the total fuel load.

The introduction of fuel load into the grassland fire danger rating system (McArthur 1977; Purton 1982) was an attempt to provide a practical solution to the situation where grasses are absent or eaten out after prolonged droughts (a condition common in the arid zone of the country) where it did not seem sensible to forecast high levels of fire danger when fires would not spread because of lack of fuel. However, the different results of the MkIV and MkV fire danger meters created confusion because fuel load was incorrectly assumed to directly influence ROS.

It may be feasible to introduce fuel load into a fire weather warning system when it can be reliably estimated on a regional basis using grassland production models (Dawson *et al.* 1991) or estimated directly from satellite imagery. However, we consider it to be more practical to calculate fire danger assuming a standard grass cover and calculate fire spread for specific fuel types using the new equations. If continuous fuels are absent local arrangements can be made concerning fire restrictions and suppression preparedness.

Fuel height Fuel height has some effect on ROS (Cheney *et al.* 1993). The effect of fuel height on suppression difficulty is not known quantitatively but can be described qualitatively. Tall standing grasses produce fires with high flames and high radiation loads: these fires may be difficult to approach directly but often present few mop-up problems once extinguished. Fires in low, compacted fuel beds may be easier to approach and suppress directly, but the compacted fuel bed may remain smouldering for long periods causing problems with mop-up and reignition from wind-blown embers. The effect of fuel height on suppression difficulty will vary widely in local areas and cannot be easily accounted for in a regional fire danger rating system.

Fuel curing New ROS equations (Cheney *et al.* unpublished) use a sigmoidal function between ROS and curing state between 50 and 100 per cent. In the MkIV GFDM, FDI increases exponentially with curing state which may well reflect changes in suppression difficulty across the landscape.

Fuel state expressed as a fraction of fully cured grass is rarely uniform over large areas early in the season because there are large differences in soil moisture between ridge-tops and gullies. Conditions can arise when grasses on ridges are fully cured and burn rapidly while grasses in gully locations are less then 50 per cent cured and will not burn. A satellite estimate of curing state based on the greenness index of 1 km square might produce an integrated curing value of 75 per cent in undulating topography. Fires will not spread uniformly but will spread rapidly on fully-cured ridges and be stopped by the green gullies. Under these circumstances suppression is greatly assisted by natural barriers. It is only when the landscape is more than 90 per cent cured that there is potential for wide-spread and devastating grass fires.

As a result fire controllers should not expect accurate predictions of fire spread across the landscape before grasses are fully cured even though the new equations may give an accurate prediction of ROS in a uniformly cured pasture.

Wind speed The new ROS equations propose a direct relationship been ROS and wind speed above a minimum threshold level. It may be that rates of spread do not continue to increase directly with windspeed at very high wind speeds (> 70 km h^{-1}). However, there is no experimental evidence to support the contention that rate of spread slows down under high winds and we consider the interpretation of field data by McArthur (1968) to be unreliable.

The exponential relationship between FDI and wind speed used in the MkIV GFDM may also be a reasonable description on the effect of wind on suppression difficulty. There is little doubt that, due to spotting, fanning smouldering combustion, and erratic behaviour, fires at high wind speeds are more difficult to suppress than fires at low wind speeds, and it well may be that suppression difficulty increases exponentially as wind speed increases.

In grass fires, the effect of fire breaks on the progress of the fire needs to be considered. At relatively low wind speeds, the potential ROS may be high in continuous fuels, but fire-spread across the landscape can be easily slowed, stopped or held up by narrow barriers such as roads and creeks. As wind speed increases fires will breach progressively larger breaks by either flame contact or by blown embers (Wilson 1988) and at very high wind speeds there will be little if any retardation of rate of spread.

CONCLUSION

We recommend that fire danger and fire spread be considered and calculated quite separately. This will allow revision of fire spread equations to be made without altering the fire danger rating systems.

Fuel load has no effect on ROS and its effect on suppression difficulty is complicated by questions of fuel continuity, fuel height and fuel compaction. Therefore, fuel load cannot be easily built into a general fire danger rating system. Use of McArthur MkV GFDM and the Purton modification of the MkIV GFDM should be discontinued.

The McArthur MkIV grassland fire danger rating system has been used successfully for three decades to provide a guide for levels of preparedness and public warning in the pastoral areas of most States of Australia. The functions used in this meter are appropriate for estimating suppression difficulty and fire authorities should continue to use this system to provide regional fire danger or fire weather warnings. The scale should be open-ended to reflect increasing suppression difficulty at very high wind speeds.

The potential rate of spread in continuous grassland fuels can be calculated for three pasture conditions: (i) undisturbed pastures or natural ungrazed grasslands, (ii) grazed or close-mown pastures, and (iii) eaten out pastures. Local adjustments to fire restrictions and suppression arrangements can be made on the basis of these calculations. The potential rate of spread in all three pasture types is likely to be higher than the average spread rate across the landscape, when wind speeds are low and before the landscape is fully cured.

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The orchestra grows! Two new fire models

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ABSTRACT

Fire models will never be exact predictors of fire spread, fire behaviour or fire danger. Fires burning in the open in natural fuels are much too complex to be modelled accurately. Nevertheless, fire models are important and there is a need for a suite, ensemble or orchestra of models which the fire manager and the community can use to meet their particular needs.

This paper reviews some existing models and outlines two models still in their developmental stages that could assist in fire and environmental management. The first is a 'synoptic' model to assist in forecasting possible 'fire activity' in Mallee shrublands.

The second is totally different in character and is an attempt to model meso-scale meteorological parameters. It treats the fire as an input to the fundamental atmospheric equations. The model was developed at the National Centre for Atmospheric Research and it shows promise for predicting fire spread of high intensity wildfires.

INTRODUCTION

In what must be his earliest written comments on fire behaviour studies, Alan McArthur in 1955 wrote that the objectives of his studies on fire behaviour (in *Pinus* species) were....

1. To determine factors which control the rate of spread of fires, and
2. To determine the relative importance of each significant factor.

Such a fundamental study to provide a basis for -

(a) Preparation of a fire danger rating system and so provide a more accurate basis for fire weather forecasting.
(b) Determination of man power and speed-of-attack requirements.
(c) Explanation of normal and abnormal fire behaviour.
(McArthur 1955)

Not much has changed in relation to fire danger forecasting objectives in the last 40 years but a lot has changed in the tools or models available to satisfy these objectives. However, it is doubtful if anyone is completely satisfied with the fire models currently available.

FIRE BEHAVIOUR MODELS FOR AUSTRALIAN USE

No one fire model is going to satisfy all the needs of any fire manager or fire weather forecaster. The wildfire process is very complicated and we are at the mercy of the complexity of the physical and chemical processes involved. A list of the subcomponents of a comprehensive fire model is awesome and an understanding is required in many different and mostly not well understood areas. A comprehensive fire model would require sub-models to describe turbulent combustion and meteorology. On top of all that, a model is required that describes the infinitely variable biological systems, for they determine fuel structure and dynamics.

The use of fire models in the future may be very like the current use of weather models. No weather forecaster relies on only one model but on a whole suite of models. Greater or lesser weight is given to various models depending upon the forecasting range or the particular weather system, to which is added the forecasters' experience. So it may be for fire managers, there will be different models for different needs. Some models will be useful for firefighting, some for prescribed burning, some for high intensity management burns, some for domestic and rural fuel management (Packham 1989).

McArthur's Models

Among the more useful models currently available are the McArthur models of various marks (see for example, Cheney 1981). McArthur models have become such a way of life in Australia, that to some people fire danger is what is produced by a McArthur Meter. The McArthur Fire Danger Meters have tended to be overused and applied outside the range for which they were designed; for example, the dry sclerophyll model being used for both the mallee and wet sclerophyll. McArthurs models are like a large and leaky ship, too valuable to scrap and too expensive to repair. A recent study of reported rates of spread in three intense grass fires by J.C. Noble (Noble 1991) has provided a boost for the Mk4 grassland model and a fatal blow for the Mk5 version. The Mk4 meter gave good predictions whereas the Mk5 (which incorporates fuel as a rate of spread variable) under predicted rate of spread by a factor of three or more.

The 'Red Book'

The Western Australian 'Red Book' fire behaviour model (Burrows and Sneeuwjagt 1988) is based on an empirical approach as was McArthur's . In our opinion it is currently the most useful and satisfying empirical operational fire behaviour model for Australian conditions. It is flexible and continues to develop at a satisfying rate as an increasing number of fuel types are included.

When high fire intensity and crowning behaviour is incorporated into this model we should reach very close to perfection for a fire management operational tool. We fail to see why the rest of the nation continues to ignore it, especially as current software makes it very convenient to use.

Tasmanian Moorland Model

A new fire behaviour model for Tasmanian moorland fuels has been developed by Jon Marsden-Smedley of the Tasmanian Parks, Wildlife and Heritage Service. It includes a buttongrass moorland fire danger index based on fire measurements taken under a wide range of conditions. The index predicts fire behaviour and suppression difficulty in these notoriously flammable fuels. The model will be published as a Parks Service Occasional Paper in the next couple of months.

The Canadian Model

The Canadian Model (Stocks *et al.* 1988) comes closest to disproving our contention that one model is not easy enough for a nation with serious fire problems. It seems to be growing and becoming so increasingly complex that it may yet provide support for this view. However, the Canadians and others seem to like it and it appears to be a suitable choice for fire managers with the fuel types and fire regimes that suit it.

It is necessary for fire behaviour models to be designed for particular climates. The Canadian and the American models are mostly designed for areas of high rainfall and lush green forests where fuel availability is determined by fuel dryness alone. In contrast, the Mediterranean climates with frequent fires, minimum summer rainfall and little winter snow cure, require models specific for those conditions. Perhaps after the appearance of a fire tornado in Australia in the form of Marty Alexander, the Canadian model may find some usefulness in Australian pines. We should observe and emulate the wonderful presentation and product support for their fire model.

The US Rothermel Model

The current operational Rothermel model (Rothermel 1972) seems to be useful in Australia in heathlands (Catchpole 1987; Catchpole *et al.* 1993). It has been shown to be useful in a variety of US fuels (Andrews 1980) and the 'new' one with the considerable involvement of the Catchpoles may have an extensive application in Australia (as encompassed in the BEHAVE fire prediction system, see Burgan and Rothermel 1984; Andrews 1986, 1989). Whether any 'new' model finds a place in Australian fire management will depend as much on promotion, marketing and ease of use as effectiveness and accuracy.

TWO MORE MODELS

Krusel Fire Activity Model

The main original aim of McArthur and Luke as they went about their fire behaviour studies was to provide the Bureau of Meteorology with a tool that would permit the forecasting of 'fire danger'. McArthur assumed that fire danger was the same as difficulty of suppression (see for example, McArthur 1958). However, other definitions take account of risk. A whole semantic debate always follows any consideration or attempt to define the term 'fire danger'.

Krusel *et al.* (1993) considered the problem of forecasting fire danger on the synoptic scale, i.e. 1000 km or 24 hours. The purpose of the study was to satisfy the need for both private and public organizations and individuals to know the likelihood that serious fire will occur over the following daily forecasting period.

The motive for developing the model came from feelings of dissatisfaction when using the McArthur Forest Fire Danger Meter Mk5 in the Mallee areas of Vic, NSW and SA. The McArthur Meter gives high fire danger classes for almost every day during the declared fire danger period. There is a severe over-warning situation.

Krusel *et al.* avoided the problems of many different definitions of fire danger by generating the concept of fire activity.

'In Australia there is no consensus on the best definition of fire activity. Foster (1976), for example, defined 'serious' fires in terms of : duration of fire; number of lives lost; area burnt; structures lost and estimated cost of damage. Cheney (1976) catalogued bushfire disasters in Australia from 1945-1975 by: duration of fire; number of days of extreme FDI; area burnt; lives lost; type of property damage; and value of damage (Consumer Price Index adjusted). Newspaper reports tend to focus on the loss of life, damage to property and on the magnitude of the suppression effort involved.'

In response to this confusion Krusel *et al.* introduced the concept of fire activity as a function of the number of fires recorded on a day, the total area burnt, the number of persons recorded as fighting the fires and the number of units attending. The indicators were put into categories and fire activity was the average of the categories of the four indicators equally weighted. Table 1 gives the criteria that were used to categorize the indicators of the activity.

The methodology was to determine the meteorological and fuel parameters that best predicted fire activity. Correlations were determined from nine years of data in the Victorian Mallee shrubland and then tested on the next two years of data.

Table 2 provides the selection criteria for estimating the fire activity categories from various meteorological parameters. The category that is determined is the highest category for that particular parameter.

Table 3 shows the efficiency of various meteorological parameters as predictors of fire activity and the number of days over a period of nine years on which a fire weather warning would be issued.

It turns out that the McArthur model provides only a 1.6 per cent probability of correctly predicting a day of high fire activity. By combining a variety of meteorological parameters the probability of detection can be made some three times better. Whilst a 4-5 per cent probability of detection does not sound impressive, the use of the new technique would cut down the number of days on which fire danger is warned by two thirds.

In another approach using these data Dowe and Krusel (1993) constructed a minimum message length decision tree. This alternative form of analysis has produced very similar results. The tree is shown in Figure 1 and the symbols used are indicated in Table 4.

Krusel has extended this study to the dry sclerophyll fuels where the McArthur Forest Fire Danger Index and the Krusel models have been shown to have a similar performance (Krusel *et al.* 1993). This is a fortunate result that confirms that McArthur has a predictive capacity in the fuels in which it was intended and that the Krusel method is a useful fire predictor.

TABLE 1

Criteria for definition of fire activity categories in the Victorian Mallee.

CATEGORY	PERCENTILE	AREA (ha)	No. OF FIRES PER DAY	PERSONS ATTENDING	FIRE UNITS ATTENDING	No. OF DAYS SELECTED
7	>99.0	14001+	7+	180+	32+	6
6	<=99.0, <97.5	5051-14000	5-6	74-179	16-31	9
5	<=97.5, >95.0	2001-5050	4	53-73	8-15	31
4	<=95.0, >85.0	46 - 2000	3	30-52	4-7	79
3	<=85.0, >75.0	9 - 45	2	21-29	2-3	154
2	<=75.0, >50.0	2-8	1	11-20	2	264
1	<50.0	0-1	1	0-10	1	336
0		0	0	0	0	2353

TABLE 2

Selection criteria for high activity days, 1979-1988.

METEOROLOGICAL VARIABLE	CATEGORY 5 RANGE	CATEGORY 6 RANGE	CATEGORY 7 RANGE
Maximum temperature (°C)	26 - 44	29 - 46	38 - 43
Days Since Rain	0 - 38	3 - 33	6 - 19
Keetch-Byram Drought Index (mm)	86 - 192	109 - 173	119 - 168
Wind Speed (km h^{-1})	0 - 48	6 - 44	0 - 37
Relative Humidity (%)	6 - 43	3 - 34	[illegible] - 24
McArthur Grass Fire Danger Index	1 - 191	4 - 118	2 - 108
McArthur Forest Fire Danger Index	16 - 93	26 - 79	28 - 71

TABLE 3

Ability of various parameters to discriminate high fire activity days based on nine years of data, 1979-1988.

METEOROLOGICAL VARIABLE(S) USED AS PREDICTOR	NUMBER OF DAYS SELECTED	PROBABILITY OF CORRECTLY PREDICTING A HIGH FIRE ACTIVITY DAY (%)
Maximum temperature (°C)	1270	3.6
Days Since Rain	3155	1.5
Keetch-Byram Drought Index (mm)	2871	1.6
Wind Speed (km h^{-1})	3215	1.4
Relative Humidity (%)	2181	2.1
All Meteorological Variables	1102	4.2
McArthur Grass Fire Danger Index	3171	1.5
McArthur Forest Fire Danger Index	2639	1.7

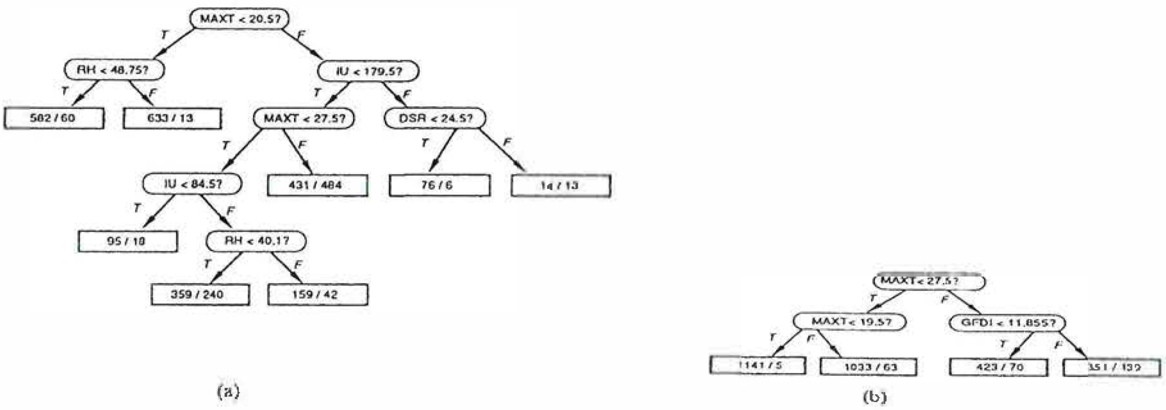

Figure 1. MML decision trees developed from the training data using all of the meteorological variables for predicting (a) 'no fire' vs 'fire', and (b) 'low' vs 'high' fire activity.

TABLE 4

Description of symbols used. The calculations typically use observations taken at 1500 hours (3.00 p.m.), unless stated otherwise (from Dowe and Krusel 1993).

SYMBOL	DESCRIPTION
RH	Relative Humidity (%)
T	Air temperature (°C)
WSP	Wind speed in the open at 10 m (10 min average)(knots)
IU	Keetch-Byram Drought Index (mm of water required to return soil to field capacity)
DSR	Days since measurable rain (measured at 9.00 a.m.)
RAIN	Amount of precipitation to 9.00 a.m.
MAXT	Maximum daily temperature (°C)
P	Air pressure at mean sea level (hPa)
TW	Wet Bulb Temperature (°C)
C	Degree of grass curing (%), assumed to be 100% (0% = fully green grass, 100% = dead grass)
D	Drought factor as on the McArthur Forest Fire Danger Meter
FFDI	McArthur Forest Fire Danger Meter
GFDI	McArthur GrassFire Danger Meter

Even if the probability of detection of a high fire activity day is not great it still has considerable economic benefits to cut down the number of false alarms. Forecasts of fire activity do not negate the forecasting of potential fire behaviour which so far has been the only basis on which fire weather warnings are issued.

The best combination of meteorological parameters explains only a small proportion of fire activity. The Forest Fire Danger Index and the Grass Fire Danger Index both do worse. The probability of ignition or the spatial occurrence of ignitions are most likely the major factors in determining fire activity in mallee fuel. McArthur and others have, from time to time claimed that the Fire Danger Index is an indicator of the probability of fires starting. There is an assumption that there is always an adequate number of potential ignitions. One curious fact (significant at the 95 per cent level) in the Krusel *et al.* study is that the declaration of a total fire ban day does not decrease fire activity in the study area.

It may be that predicting ignition probability is beyond landscape fire scientists, or might take us into such chaotic fields as human behaviour where we do not feel comfortable. However, if an analysis of ignition sources is made, we will find that there is one source of purely meteorological origin, e.g. lightning. It appears fairly certain that Australia will at last acquire a lightning detection system and our skills are then sure to improve.

Some ignitions are accidental and as we could expect, the ignition sources have a fairly constant rate of occurrence. The occurrence of wildfires is then determined by the conditions of the fuels and the weather. A case in point is the ability for fires to start along freeways under extreme fire danger conditions, but rarely when the fire danger is lower. The number of potential ignition sources could be expected to be dependent only on the use of the freeway.

However, other ignitions are the results of a conscious act of either carelessness or malevolence and may be harder or even impossible to forecast. The question of exciting arsonists with a fire ban needs to be investigated with a rigour and skill that is the trademark of good experimental psychology.

Meso-Meteorological Models Applied to Wildfire

The basis of meteorological modelling is the fundamental equations of fluid dynamics. A solution of these equations for each spot in the earth's atmosphere would provide perfect weather forecasting. The equations cannot be solved analytically and thus computer solutions are sought to a grid cell approximation to the atmosphere to provide weather forecasting. Meteorological models are a big industry and every self respecting nation must have one or access to one. Within Australia we do very well having a few of our own as well as access to European, British and American models on a twelve-hourly basis. Most meteorological models are built on the synoptic scale, i.e. a few thousand kilometres, and typically provide solutions for points in the atmosphere on a grid of 75-150 km. Not much use for studying the interaction of fires on the weather.

Meso-models operate at a much finer scale from about 10 m - 20 km and may turn out to be particularly valuable for studies of wildfires.

In principle, the set of fundamental equations should be able to encompass the heat and mass release of a fire front and be able to predict such things as height of convection, windfields around a fire front, and rate of spread.

The most common approach to modelling fire spread is to assume that the fire is driven by radiation processes that heat the fuel to some combustion temperature. We suggest that the failure of most of these models is a fair hint that the approach has limitations. We suggest further that fires are primarily convective processes and that it is possible that modified meteorological models could be quite useful in modelling fires as long as we can dynamically couple the fire and meteorological physics correctly. An international co-operative project is underway to attempt to do just that.

Clark (Clark 1977; Clark and Farley 1984) has developed a complex non-hydrostatic meso-model. Non-hydrostatic means that vertical air accelerations are explicitly treated in the model equations. The computing requirements for such models are large and as a result they are expensive.

The Clark model has been successful in demonstrating the mechanisms of atmospheric processes involving strong buoyant forces. The severe downslope windstorm is an example, a model output is shown in Figure 2.

The gusts associated with these types of windstorm have a characteristic periodicity of several minutes. The gusts descend onto a particular zone downwind of the Rocky Mountain range. They are of an intensity capable of causing considerable property damage and are now know to result from the breaking of atmospheric waves which had been generated by the steep upwind topography.

Work is proceeding on including a bushfire component within the Clark model. A prototype has been applied to the Tasmanian environment to get a feel for its ability to model some of the wind field peculiarities of that very 'bumpy' State. A simple model was developed of the heat and mass transfer (at meso-scale) in a high intensity crown fire under the Hobart 1967 conditions. The simple model was too gross and blew up the meso-model. The fire model assumed that all the surface energy was used in drying the canopy before canopy ignition. It is probable that only some 25 per cent of the heat produced by ground combustion is used in canopy drying.

The spread rate was far too high, probably a factor of twenty, prompting a search for a better physical model of fire and a negative feedback mechanism.

The initial attempts have used the McArthur model as a predictor of fire behaviour, the meso-model supplying the meteorological parameters for insertion into the McArthur Forest Fire Danger Meter. It is hoped that this empirical component can be relaxed over the next stage of model development.

If fire is a convectively driven phenomenon then a convective parameter based on vorticity dynamics may well be needed for predicting both rate of spread and energy release.

The Clark meso-model is currently a research model and one that is usually run on a supercomputer. On the Cray Y-MP it runs about nine times real time and so is unlikely to ever become an operational model in its present form. Modelling technology is improving very rapidly and it is expected that in a few years a real time workstation model may be available.

The work is complex and confusing but the results look promising.

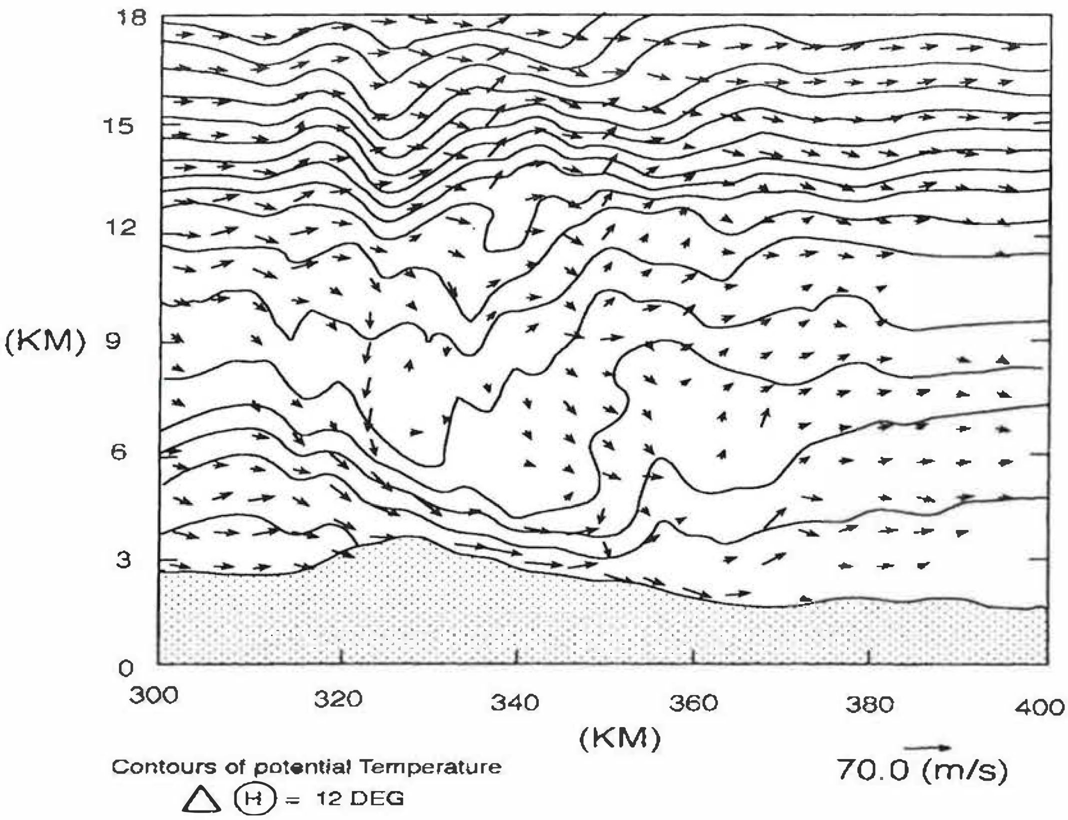

Figure 2. Downslope windstorm Rocky Mountains.

CONCLUSION

There is in existence a range of fire models of different types, all of which can be useful for fire management. They include the Western Australian Red Book model that could be put to much more use than it has been so far.

The Krusel model for fire activity forecasting in Mallee and dry sclerophyll fuels should be given a careful testing in either its simple or decision tree form.

An exciting possibility is the application of non-hydrostatic meso-meteorological models to predicting fire weather interaction including convection and even fire behaviour.

The times ahead are exciting and we must maintain the momentum.

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Time dependence of temperature above wildland fires

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ABSTRACT

A statistical description is presented for the time dependence of temperature at various heights above a moving wildland fire. The model was developed using nonlinear least-squares curve fitting on experimental data from fires in shrubland fuel complexes and is based on classical theory with empirical modifications (where necessary) to suit the fuel types being studied. Relative to a stationary observation point which is overtaken by a spreading fire, the temperature-time history can be partitioned into two distinct regions:

(i) As the fire approaches, the rapid temperature rise above ambient is modelled by a Gaussian curve, having only one free parameter β to describe its steepness. Maximum temperature rise in measured data generally occur within 60 seconds.

(ii) As the fire recedes, the temperature falls comparatively slowly, with the fastest rate determined by simple Newtonian cooling (again described by one free parameter, γ. In practice, residual burning in larger components of the fuel bed results in a long 'tail' so that γ becomes an effective value.

Fire effects upon vegetation are discussed and a means of comparing lethal exposure experiments in the laboratory with wildland fire temperature-time curves is detailed.

INTRODUCTION

In studying the ecological impact of wildfires, it is desirable that the temperatures to which plants are subjected and the durations of those temperatures are known. In a related paper, Weber *et al.* (this publication) present a model for the variation of maximum temperature with height above ground for free-burning wildland fires. This new study considers the time dependence of the temperature rise, for any height above a given point on the ground, as a firefront approaches, passes over and then recedes. In particular, the time at temperature, for any given temperature rise ΔT, has a major impact on the survival probability of vegetation. Martin *et al.* (1969) have presented temperature lethality curves for leaves and seeds. We later relate our findings to theirs and detail a method for determining lethal temperature exposures, a long-standing problem.

MATHEMATICAL THEORY

Temperature Rise

From inspection of many sets of experimental data, the temperature at a given height rises abruptly from ambient to a maximum value, naturally suggesting a plume-like, Gaussian-shaped dependence as suitable and appropriate given the level of accuracy inherent in thermocouple data. This is supported by the theory of Yih (1969), who modelled the temperature rise ΔT for a large height z above and for a horizontal distance x away from the plume due to a stationary line source of heat

$$\Delta T = kI^{2/3} / z \exp(-x^2/\alpha^2 z^2) \tag{1}$$

where k is a proportionality constant, I is the line fire intensity and x is the horizontal distance. A large height is necessary as the theory of Yih (1969) assumes a line heat source. The quantity α is an entrainment constant approximately equal to 0.16 (Lee and Emmons 1961). A wildland fire moving at a constant rate of spread U approximates a line source at position x=Ut so that we may recast equation (1) as

$$\Delta T = A / z \exp(-t^2/\beta^2 z^2) \tag{2}$$

where $\beta = \alpha/U$, $A = k\, I^{2/3}$ and $t = 0$ at the time of maximum temperature. (Note that this equation with t=0 gives ΔT_{max} versus height and has been used by Thomas (1963) and Van Wagner (1973).) Thus t is negative prior to the arrival of the firefront.

Temperature Fall

From simple Newtonian theory, the cooling rate of a hot object is proportional to its temperature elevation above ambient

$$dT/dt = -\gamma (T - T_a) \qquad (3)$$

where γ is a constant for a given fuel. Integration of equation (3) yields the following expression for ΔT as a function of time

$$\Delta T = B \exp(-\gamma t) \qquad (4)$$

The obvious boundary condition is that B is equal to ΔT_{max} and thus $B = A/z$ necessarily, where A/z was defined for the temperature rise. This behaviour should apply exactly to fires where the combustion residence time is negligible. From a fluid-mechanical analysis of the cooling phase, assuming idealized Newtonian cooling; a value for γ of order 0.1 is obtained. As seen later, this is much greater than the fitted values and is thus equivalent to much faster cooling than occurs in practice. Real wildland fires leave in their wake a trail of partially-combusted fuel, from smouldering ash through glowing coals to actively flaming logs. Therefore, γ obtained from a least-squares fit to data from such fires will be an effective cooling constant.

FITTING EXPERIMENTAL DATA

Temperature-time Dependence

A series of experiments has been performed in Ku-ring-gai Chase National Park, near Sydney, and at the CSIRO Kapalga Experimental Station in Kakadu National Park, Northern Territory. The Ku-ring-gai fuels were shrubby and varying in depth from 0.5 m to about 2 m (Weber *et al.* this publication), while the Kakadu vegetation consisted of mainly long dry grass, an intermittent shrubby understorey and eucalypts to about 15 m maximum height, Moore *et al.* (this publication).

Data from seven Ku-ring-gai fires and 20 Kakadu fires, obtained over a three-year period, were analysed. The temperature data were obtained from arrays of shielded Type K (chromel-alumel) thermocouples which were mounted on an aluminium mast and spaced between ground level and 9 m. For details of the Ku-ring-gai experiments the reader is referred to Bradstock and Auld (1994). In this paper we present only results at the 3.5 m and 6.0 m level above ground where the 1/z plume theory is more likely to apply and the moving fire appears more like a line source, so that equation (2) should be valid. Figures 1 and 2 show the results of fitting equations (2) and (3) to one of the Ku-ring-gai fires at 3.5 m and 6.0 m above ground level respectively. Both curves were constrained to pass through ΔT_{max}. Note the rapid rise in temperature to maximum (within 60 seconds) after the fire's onset, followed by a long, slow fall in temperature over more than 300 seconds following maximum temperature. However, at large positive values of time, temperatures tend to remain elevated above those predicted by the Newtonian cooling model. This behaviour is reflected in the fitted values for the cooling parameter which were 0.013 and 0.009 at the 3.5 m and 6.0 m levels respectively. These are an order of magnitude smaller than the theoretical Newtonian value of about 0.1. The fluctuations in ΔT which occur during cool-down probably correspond to small-scale flaring of fuel elements and also to slow fluid-mechanical pulsations connected with air entrainment.

Figures 3 and 4 depict the temperature-time history for a Kakadu fire, again for 3.5 m and 6.0 m above ground level. The scaling is quite different to that of the Ku-ring-gai fire yet the same qualitative behaviour is evident. Note that, as in Figure 2, the model *under-predicts* the decline in temperature as it is unable to accommodate prolonged smouldering or persistent localized combustion after the firefront has passed. The fitted value of γ was 0.016 at 3.5 m and 0.013 at 6.0 m. In both fires the fit around the peak is very good.

Total Time-above-temperature

The simple approach to predicting thermal death of plant materials in fires would be to take the length of exposure above certain temperatures and compare the results with those from constant temperature exposure in a furnace or water bath. Given experimental temperature data or fitted curves such as those in Figures 1 to 4, the total time for which a given value of ΔT is exceeded may be read directly off the graph. However, with a little algebra, we may express time above ΔT as a function of ΔT as follows:

From equation (2) the time above ΔT during which the temperature is rising is

$$t_r = \beta z\, (\ln(A/z\, \Delta T))^{1/2} \qquad (5)$$

From equation (4) the time above ΔT during which the temperature is falling is

$$t_f = (1/\gamma)\, \ln(A/z\, \Delta T) \qquad (6)$$

Therefore the total time above ΔT is

$$t_{tot} = \beta z\, (\ln(A/z\, DT))^{1/2} + (1/\gamma)\, \ln(A/z\, \Delta T) \qquad (7)$$

Fitted values of A, β and γ were inserted into equation (7) and the resulting curves are shown in Figures 5 to 8.

It is also possible to obtain an estimate of time above ΔT directly from the thermocouple data. This is fraught with problems as it requires an interpolation (by eye) between data points. However, in order to evaluate our method of fitting temperature-time curves, it seemed appropriate to compare the two methods of finding time above ΔT. Hence the data points on Figures 5 to 8, which exhibit the same trend as the curve from equation (7).

USING TIME ABOVE T TO ESTIMATE DEATH OF VEGETATION

Laboratory measurements of temperature-time exposures which cause leaf death usually come from bathing the sample in a constant temperature and measuring the time till death. This is not a true representation of the temperature exposure in a fire, due to many factors, including the variability in thermal environment associated with wildland fires and the different thermal properties of water and air, but, for obvious reasons, is the convenient experiment to perform. It provides curves like those in Martin *et al.* (1969), characterized by the equation

$$\ln t_d = a - b\,T, \tag{8}$$

where t_d is the time to death at an exposure temperature T.

A way in which these laboratory curves might be used together with our temperature-time relationship (equation (7)), to predict leaf death and other fire effects is given below. It is the heat flux and the ability of the vegetation to dissipate heat that governs the temperature rise of a sample. However, the flux is difficult (if not impossible) to estimate for a given fire, at a given height and time, even with temperature information. Hence, in the absence of detailed understanding of the fluid mechanics, we are forced into considering only the temperature information available to us.

We first notice that the lethal time at a constant temperature, T_{const}, will be more than the lethal time at a varying temperature, $T_{var}(t)$, where the minimum is always equal to, or greater than, T_{const}. It is then clear that we can perform a direct comparison of the time above a given temperature curve from a fire, with the death curves found in the laboratory. This provides a bound on the effects of the fire on vegetation. Namely, if the time-temperature curve is ever at a higher temperature than the death curve then the vegetation being considered will perish. However, if the death curve is always above the time-temperature curve we cannot be certain of the fate of the vegetation. These possibilities are shown together on Figure 9.

A better method to determine the fate of the vegetation, particularly in this uncertain zone, consists of the following. Divide the time-temperature curve into discrete temperature ranges and determine the time spent within a particular range. The ratio of this calculated time to the laboratory measured time to death at a representative time in the same range is then determined. This allows for both the heating and cooling phases that the vegetation is subjected to. If we assume additivity of these exposures then the sum of these ratios will give an indication of the likelihood of death. Mathematically this can be expressed as a death number, D,

$$D = \Sigma_i\, t(T_{i-1}<T<T_i)\,/\,t_d(T_i^*) \tag{9}$$

where $t(T_{i-1}<T<T_i)$ is the time spent in the temperature range (T_{i-1},T_i) and $t_d(T_i^*)$ is the laboratory-measured time to death of a representative temperature (T_i^*) in the range (T_{i-1},T_i). If the sum of these ratios, the death number, (D) is greater than 1, then death is likely, and if the sum is less than 1, then the vegetation is likely to survive. Of course, for values near 1, the outcome is still uncertain. The larger the number/closer together the discrete temperature ranges considered, and hence the narrower the temperature range, the more accurate this method will be. In the limit as the width of the temperature range tends to zero equation (9) can be rewritten

$$D = -\int_{\tau=T_a}^{\tau=T_{max}} \left((1/t_d) * dt(\tau)/d\tau \right) d\tau \tag{10}$$

which, using the model for the external temperature (equation (7)) and the model of Martin *et al.* (1969) for the time to death (equation (8)), gives the death number as

$$D = (\beta z/2 + 1/\gamma)\, e^{-a} \int_{\tau=T_a}^{\tau=T_{max}} e^{b\tau}/(\tau - T_a)\, d\tau \tag{11}$$

What is needed are the laboratory-measured time to death of the vegetation (leaves, fruit, stem) for as many different temperatures as possible to determine $t_d(T)$ and the external temperature profile either from experiments or the model detailed previously.

The fires considered here both have a maximum temperature well above 60°C and hence leaf scorch will occur at the heights in question. The effects of the fire exposure on the fruits and stems could be determined using the method outlined above. Unfortunately, at present the data needed to use equation (9) or to fit to the model of Martin *et al.* (1969) (equation (8)) to find $t_d(T)$ is not available in the literature. Recently Mercer *et al.* (1994) have used the model outlined here as the external temperature input to their model for the temperature exposure of seeds in woody fruits.

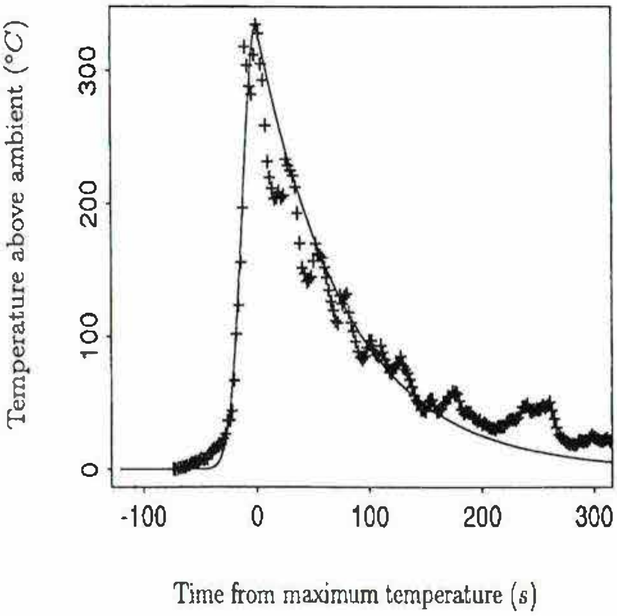


Figure 1. Temperature-time curve and data for the Ku-ring-gai fire at 3.5 m.

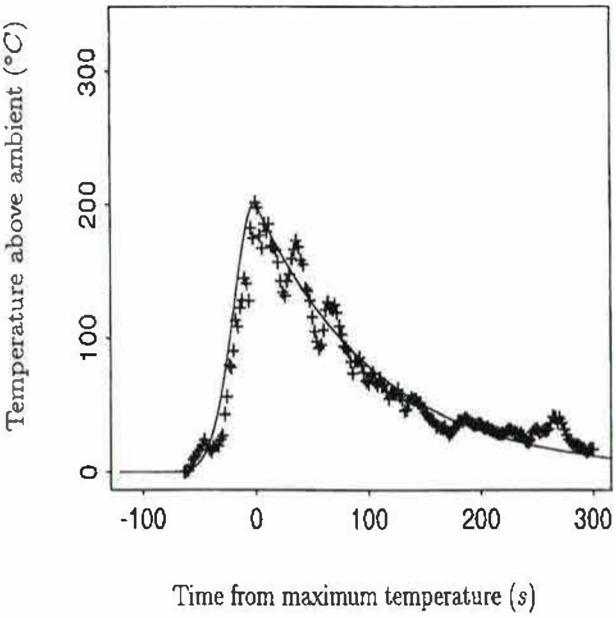


Figure 2. Temperature-time curve and data for the Ku-ring-gai fire at 6.0 m.

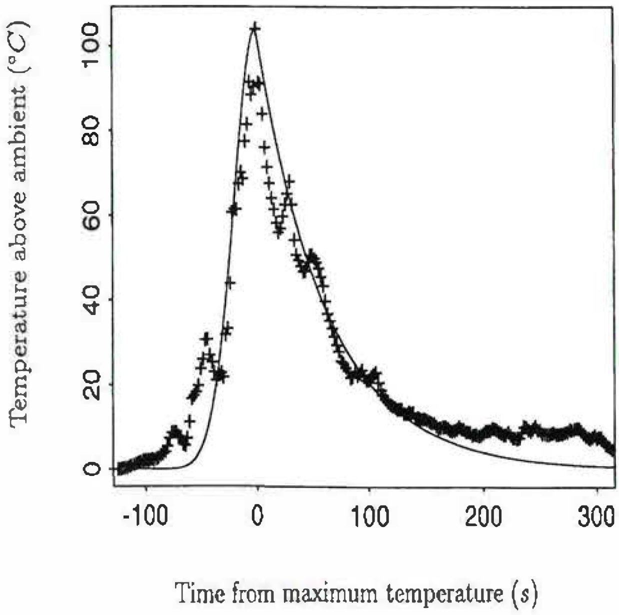
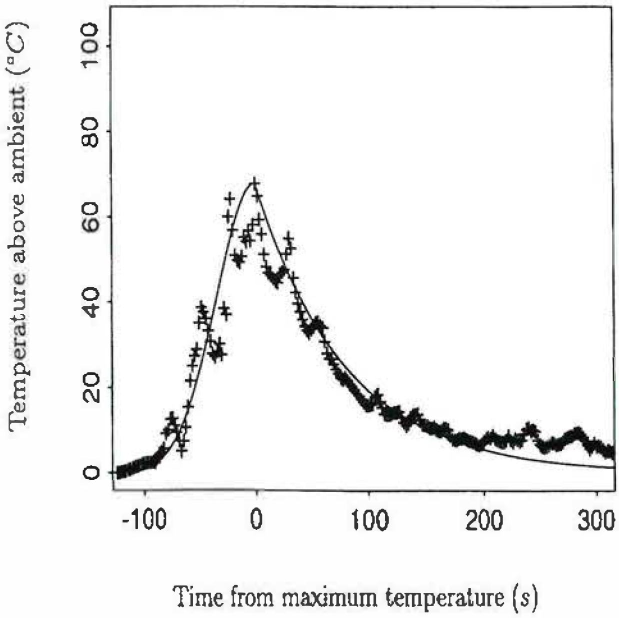


Figure 3. Temperature-time curve and data for the Kakadu fire at 3.5 m.



Figuer 4. Temperature-time curve and data for the Kakadu fire at 6.0 m.

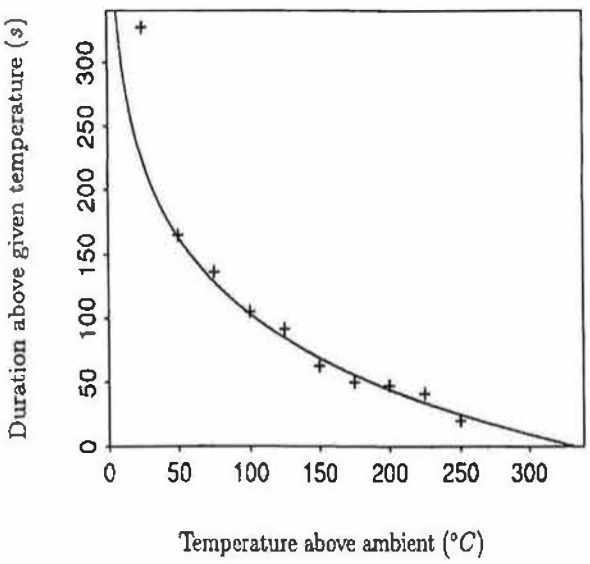


Figure 5. Duration of the fire above a given temperature for the Ku-ring-gai fire at 3.5 m.

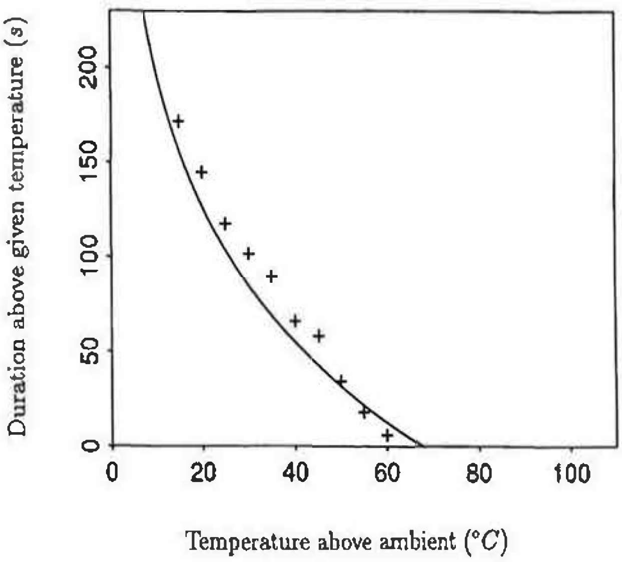


Figure 8. Duration of the fire above a given temperature for the Kakadu fire at 6.0 m.

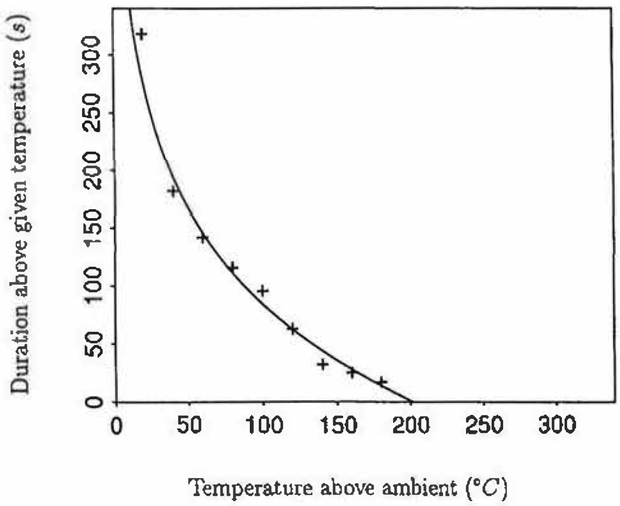
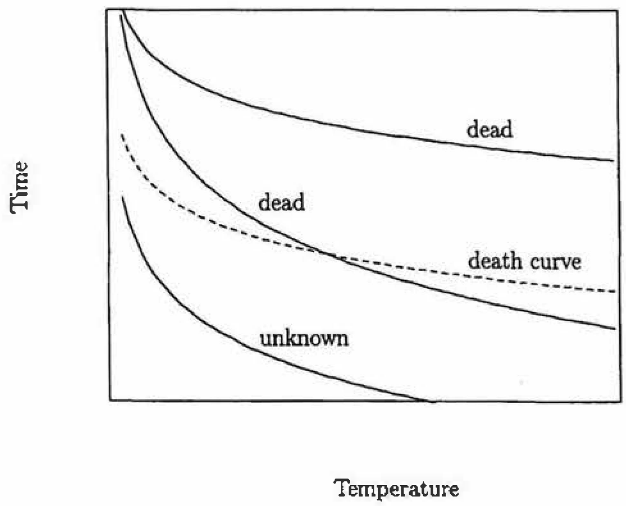


Figure 6. Duration of the fire above a given temperature for the Ku-ring-gai fire at 6.0 m.



Figuer 9. A typical death curve for vegetation and some of the possibilities for the time-temperature curves and the resultant effect on the vegetation.

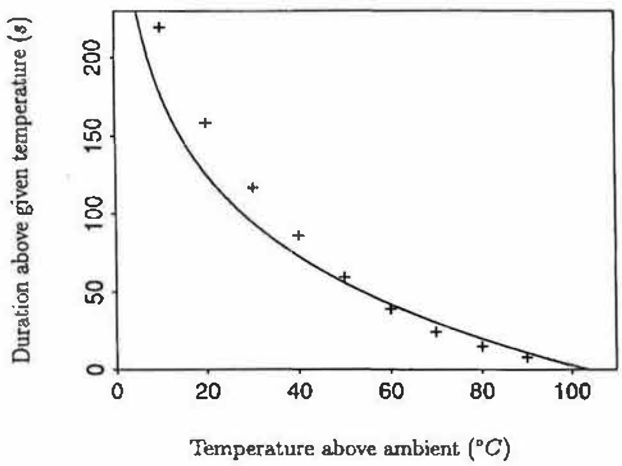


Figure 7. Duration of the fire above a given temperature for the Kakadu fire at 3.5 m.

ACKNOWLEDGEMENT

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Modelling wildland fire temperatures

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ABSTRACT

A mathematical model of the maximum temperature as a function of height above the ground, within and above spreading wildland fires, is presented and evaluated. The model is based upon classical work on turbulent plume theory with extensions to include the flaming combustion region above and within the fuel bed itself. This results in a 'three zone' model which gives an appealing qualitative description of wildland fire temperatures. Temperature versus height measurements made in shrubland fuel complexes within Ku-ring-gai Chase National Park, Sydney, New South Wales, are used to calibrate the model. The ability to include the fuel bed and flame temperatures is a significant advantage over previous models which only applied to the plume.
Keywords: Fire temperatures, Models, Australia

INTRODUCTION

Ecological studies of the impact of wildfires need to be concerned with the temperatures to which plants will be subjected. While many measurements of the temperatures obtained within and above wildland fires have been made (e.g. Trabaud 1979), no-one has been able to account for the full variation of temperature with height. Indeed, sampling heights have usually been restricted to less than a few metres. Usually it is only the plume region, above the wildfire, which has been modelled. While this may suffice for studies of crown death, e.g. Van Wagner (1973), it is insufficient for a full understanding of fire effects upon vegetation.

The model presented here is for maximum temperature reached as a function of height as a wildland fire passes. The fuel bed, the flaming region above the fuel bed and the fire plume are all included in a consistent model. Separate temperature-height functions are used in each of the three regions and we demand continuity of temperature and flux (temperature gradient) across the borders of the regions. In this way we are able to match a constant temperature region in the fuel bed with an exponential reaction-diffusion region for flaming and the classical turbulent plume for above the fire.

A knowledge of the temperature above the flames and how this is related to the flame temperature is important in the study of the impact of fire on vegetation above the flames. Issues such as leaf scorch, seed death and stem death are all reliant on a knowledge of the heat exposure of the vegetation.

PLUME STUDIES

Yih (1951) calculated the temperatures reached in a turbulent plume resulting from an idealized line source of heat

$$\Delta T = k\, I^{2/3} / z \qquad (1)$$

where ΔT is the temperature rise above ambient temperature, I is the intensity of the line source per unit length, z is the height above the line source and k is a proportionality constant. Note that equation (1) is applicable directly above the stationary source, x = 0. At other points the equation found by Yih (1951) is

$$\Delta T = (k\, I^{2/3} / z) * \exp(- x^2/\beta^2 z^2) \qquad (2)$$

where x is the horizontal distance from the source and β is an entrainment constant
(β = 0.16 according to Lee and Emmons, 1961.)

Although, strictly speaking, Yih's (1951) results apply only for a stationary line source in a quiescent atmosphere, Thomas (1963) and subsequently Van Wagner (1973) have used equation (1) in modelling the temperature rise above wildland fires. As one reaches a height far above a real wildland fire, it can be reasonably approximated as a line source and it appears to move only very slowly. Thus Thomas (1963) and Van Wagner (1973) have had some success; Van Wagner (1973) in particular, with providing a first model for crown death.

A laboratory study of stationary pool fires by Kung and Stavrianidis (1982) provides one of the best experimental tests of the applicability of the similarity analysis of Yih (1951) to fires. The experimental results show impressive agreement in the plume region. However, there is a significant region of measured temperature increases, in and around the flames, where the theoretical results are inadequate.

Our main motivation is to provide a model for the entire temperature profile for wildland fires. The two main benefits of this would be

(i) the ability to predict maximum external temperature rise to which vegetation would be subjected at any height;

(ii) a clarification of the height above a fire at which plume theory can be reliably applied.

Item (i) depends upon the combustion characteristics of fuel types as well as the fluid mechanics of the fire, and we can only provide a partial realization in this paper. Item (ii) depends solely upon the fluid mechanics and we show that our model admits an understanding of the significance of the height at which the classical plume theory becomes applicable.

Other factors which one would like to include are the movement of the fire, fire depth, wind profiles and rough terrain. However, these will not be considered in this paper.

TEMPERATURE MEASUREMENTS

Temperature measurements were made in a series of experiments in Ku-Ring-Gai National Park in NSW, Australia. The measurements involved the use of a single vertical array of sheathed Type K (chromel-alumel) thermocouples to measure temperature rises above ambient in experimental fires in heathy fuels. Fires were lit only on days of very light winds. The depth of the fuel varied greatly, from 0.5 m to 2 m, and the flames ranged in height from 1 m to 10 m. A discussion of the utility of such measurements can be found in Gill and Knight (1991). Typical results are as shown schematically in Figure 1. For clarity, a detailed report of the experiments will not be included here. Readers are referred to the detailed report on the experiments, as opposed to the temperature measurements, that can be found in Bradstock and Auld (1994).

The similarity of form, despite quite different fuel depths, is very encouraging for the development of a universal model. This was the original aim of Thomas (1963) and Van Wagner (1973), founded on the

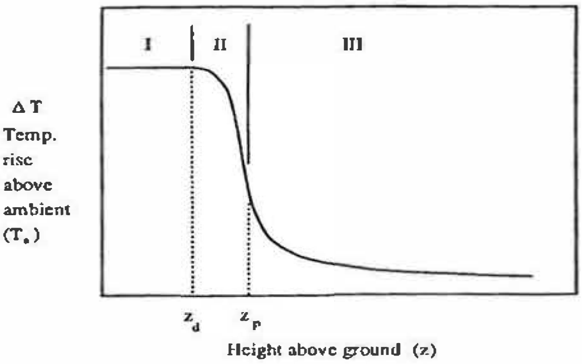

Figure 1. Typical form of curve for temperature rise above ambient vs. height above ground.

expectation that dimensional reasoning provides relationships which are scale invariant. The key problem that arose was the inability to determine a universal value for the plume constant (k in equation (1)). In our analysis, we wish to shift the emphasis away from finding universal constants. Rather, it is our expectation that the full temperature versus height profile can be understood with a 3-region paradigm. This new paradigm is itself a leap in understanding. Furthermore, it allows a comparison of fires and their potential impact upon vegetation. In the next two sections we present our paradigm and relate it to the Ku-Ring-Gai fires.

A 3-REGION MODEL

A typical curve of temperature rise above ambient, ΔT, versus height, z, can be divided into 3 regions, as shown in Figure 1.

$$\begin{array}{llll} \text{I} & \Delta T^{I} = K, & 0 \le z \le zd & \\ \text{II} & \Delta T^{II} = K\exp(-\alpha(z - zd)^2), & zd \le z \le zp & (3) \\ \text{III} & \Delta T^{III} = C/z, & z \ge zp & \end{array}$$

where K, C, α, zd and zp are constants which need to be determined. In Region I it is anticipated that the presence of combusting solid will create a constant high temperature region which extends through a height zd, perhaps comparable to the fuel bed depth. In Region II the flames mix with entrained air and an exponential decrease in temperature rise, following a Gaussian distribution, is assumed. Region III is the plume region, and extends above a height zp.

In order to reduce the number of constants which need to be determined, and to provide a smooth ΔT versus z curve, the temperature rise and the gradient will be matched across the boundaries between regions

$$\Delta T^{I} = \Delta T^{II} \quad \text{at } z=zd \qquad (4.1)$$
$$d(\Delta T^{I})/dz = d(\Delta T^{II})/dz \quad \text{at } z=zd \qquad (4.2)$$
$$\Delta T^{II} = \Delta T^{III} \quad \text{at } z=zp \qquad (4.3)$$
$$d(\Delta T^{II})/dz = d(\Delta T^{III})/dz \quad \text{at } z=zp \qquad (4.4)$$

There is a little algebra which needs to be done (see Appendix), but then these conditions determine two of the constants in terms of the other three:

$$C = K\, zp \exp(-\alpha\, (zp - zd)^2) \quad (5.1)$$

$$\alpha = 1/(2\, zp\, (zp-zd)) \quad (5.2)$$

Therefore, the model for ΔT versus z, consists of equations (3) subject to equations (5): and there are three remaining parameters to be found. These are:

(i) K, the maximum temperature reached anywhere. It may be possible to estimate this from a combustion calculation, assuming a certain proportion of total heat generated is lost to the atmosphere.
(ii) zd, perhaps related to fuel bed depth or zone of persistent flame.
(iii) `zp, perhaps related to the height of the flames in the zone of flame flickering.

It is valuable to have these guiding roles for K, zd and zp when one comes to fit wildland fire data. A detailed survey of ΔT versus z data from wildland fires is required to fully justify these guiding roles for K, zd and zp. In this context one should note the extreme paucity of published data which uses thermocouple array, or other temperature measurement means. The most detailed studies known to the authors are Tunstall *et al.* (1976), Van Wagner (1975) and Williamson and Black (1981), none of which, in their current form, can be compared with our three-zone model.

FITTING EXPERIMENTAL DATA

In order to determine the paramters in the 3-Region model, seven of the experimental fires were studied in detail. The 3 model parameters were first determined approximately by simply viewing the T versus z plots of the experimental data.

This instantly provided a smooth curve which gave a good fit to the data. Further refinement using a simple least squares routine in order to minimize the error was then done. This usually provided a small improvement to the original fit. In Figure 2 and Figure 3 we present the results of curve fitting from two of the experimental fires. For the fire presented in Figure 2 the fuel bed was quite deep and the flames quite high, hence we chose K = 700°C, zd = 1.5 m and zp = 2.15 m. For the fire presented in Figure 3 the fuel bed was much shallower, although the flames were of a similar size, hence we chose K =790°C, zd = 0.25 m and zp = 2.00 m.

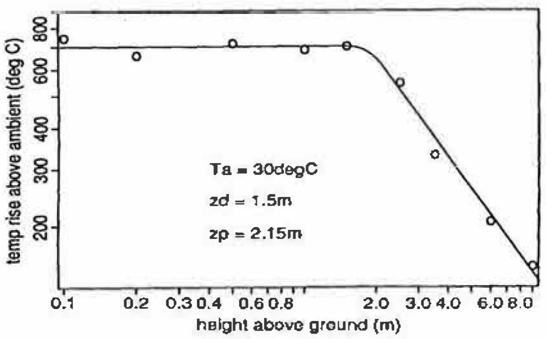

Figure 2. Log-log plot of temperature-rise profile for a deep fuel bed.

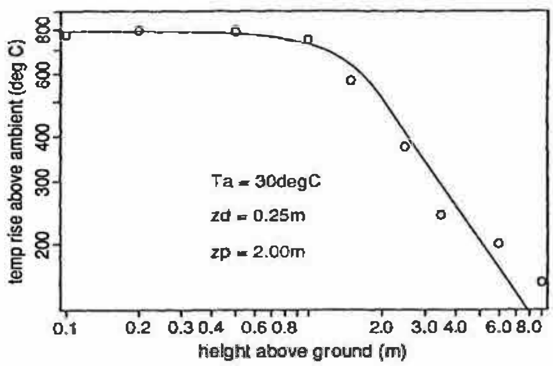

Figure 3. Log-log plot of temperature-rise profile for a shallow fuel bed.

The ability of the curves to fit the experimental data with a minimum of fuss is most impressive. It should be stressed that the model is not yet predictive. Indeed, detailed measurements in a given fuel type would be required to calibrate the model prior to using it in a predictive sense. Environmental factors such as wind, humidity and fuel moisture would also need to be taken into account. Despite this, even prior to any calibration, the results provide an insight into where the fire plume region begins. Namely, for the heath communities in Ku-ring-gai, NSW, it would seem that 2 m is the minimum height at which plume theory can be successfully applied.

DISCUSSION

The three regions identified here could match those identified by McCaffrey (1979) for fires burning natural gas above ceramic plates in the laboratory, *viz.* a continuous-flame region, an intermittent flame region, and a plume region. In wildland fires the fuel is solid - unlike the fuel in McCaffrey's fires - so it is possible that our Regions, especially Region I, may be related to fuel bed characteristics such as depth. Region I is effectively missing in shallow fuels, such as litter (Stott 1986, author's unpublished data), but readily identified in deeper fuels, such as shrublands (present work, Trabaud 1979) and tall grasses (Tunstall *et al.* 1976). Therefore, the situation described in this paper is the more general one.

FURTHER WORK

The main avenue for further work at present is to include the dependence upon horizontal distance from the fire, an analogue of time. For the plume region, this is already taken care of with equation (2). However, in the other two regions the most versatile x-dependence is yet to be determined.

One could pursue laboratory work, to determine how the variables introduced in equation (3) relate to fuel bed depth and heat release rate. This would require a large combustion chamber so that there is negligible interaction between the plume and the walls and ceiling.

Other possible inclusions are fire speed and flame depth which are related to x-dependence, and of course wind profiles. All the present work is for light winds, which is a severe limitation.

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NOMENCLATURE

- C constant
- K constant
- k constant
- I line fire intensity
- ΔT temperature rise above ambient
- x horizontal distance
- z height above ground
- zd height related to the fuel bed depth
- zp height related to the start of the plume
- α constant
- β entrainment constant (0.16)

APPENDIX

Of the matching conditions, equations (4.1) and (4.2) are automatically satisfied by our choice for equations (3.1) and (3.2). This leaves equations (4.3) and (4.4) to be satisfied. Equation (4.3) yields

$$K \exp(-\alpha (zp - zd)^2) = C / zp \qquad (A.1)$$

while equation (4.4) yields

$$-2 \alpha (zp - zd) K zp \exp(-\alpha(zp - zd)^2) = - C / zp^2 \qquad (A.2)$$

Equation (A.1) can be immediately rearranged to give equation (5) in the text. Into equation (A.2) we substitute equation (A.1) to give

$$-2 \alpha (zp - zd) C / zp = - C / zp^2 \qquad (A.3)$$

which is rearranged and simplified to give equation (5.2) in the text.

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The second generation United States Forest Service fire behaviour model (abstract only)

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ABSTRACT

A comprehensive series of laboratory fires are being burned in the wind tunnel at the Intermountain Fire Sciences Laboratory in Missoula, MT. Together with field data from experimental and wildfires, information from these fires will be used to formulate and test the US Forest Services second generation fire behaviour model which is currently being developed.

The fuels used in the laboratory fires are excelsior (wood shavings), pine needles, and 6 mm diameter sticks. Most fires are in homogenous fuel beds, but a series of fires have been burned in mixed excelsior and sticks, to compare fire behaviour in mixed fuel with behaviour in component fuels.

In conjunction with Dick Rothermel of the US Forest Service, Ted Catchpole and I are developing a firespread model based on radiative and convective heat transfer from the flame to the unburned fuel. Each fire is instrumented with thermocouples, pitot tubes and radiometers to measure the factors influencing fire behaviour. Currently the model appears to predict fire spread well if these factors are known. In order to construct a predictive model it remains to model these factors in terms of fuel and environmental factors that can be measured just prior to a hazard reduction burn or wildfire.

I will talk about progress on the model to date, the experimental results obtained so far, and about preliminary attempts to test current flame structure models which are needed to describe the heat source.

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Fire modelling and fire weather in an Australian desert

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ABSTRACT

Hummock grasses form a discrete fuel for landscape fires in the vast arid and semi-arid regions of Australia. For fires to spread in such discrete fuels, the flames need to be long enough to cross gaps, under sufficient wind to have large tilt angles, and impinge on the next hummock long enough to ignite it. Wind speed, discrete-fuel loadings, fuel moisture contents and gap-size distributions are key characteristics for fire-spread modelling in these fuels. Present models derived in the arid region do not have universal application.

Once a model is formed, formal prediction of fire spread requires a three-stage process: (i) a domain analysis for the applicability of the inputs to the fire-spread model; (ii) a likelihood-of-any-spread analysis; and, (iii) application of a spread model to predict rate of spread of the headfire.

That direct inputs, such as fuel moisture, are not available for models on a routine basis creates problems of prediction. For fully cured fuels, fuel moisture may be expected to be a function of fuel temperature, fuel humidity, and wind speed near the ground; these variables are not directly available so they, too, must be predicted. With each extra step in the estimation of inputs, further errors in spread prediction are likely to arise.

At Rudall River National Park in north-western Western Australia, fire weather may reach extremes. Temperatures are often over 40°C, dew points drop to – 37°C; and winds, unattenuated by trees, reach 50-60 km h^{-1} at any time of the year. Fuels may appear completely dead while gaps between plants may extend a metre or more.

Appropriate fire-weather indexes for the arid region need to be determined independently of fuel considerations and should not be tightly linked with fire behaviour because of the great spatial variety found in fuel condition. Any one-step index linked to fire behaviour is likely to fail because several variables need to reach threshold levels before spread will occur.

INTRODUCTION

Hummock grasses are the main fuel type over thousands of square kilometres of arid Australia. Fire intervals there are thought to be from 5 to 50 years (Walker 1981). However, there have been few studies of fuels, fires or weather. In this paper, we introduce a number of problems, submit some new ideas, and present pertinent data and experiences relating to the spread of fires in hummock grasses. We examine the context of fire spread in and near the 1.6 million hectare Rudall River National Park (Australia's second largest national park) in arid north-western Australia. Because the relief of the landscapes are generally subdued we ignore terrain. We consider only fuels, fire weather (in the broadest sense) and the spread of fires.

DISCRETE HUMMOCK-GRASS FUELS

Hummock grasses are the predominant fuel of the arid zone; they are a unique Australian growth form in which:

> 'Each plant branches into a great number of culms which intertwine to form a hummock and bear rigid, involute, pungent leaves representing a serried phalanx to the exterior.' (Beard 1981)

Hummock grasses belong to the genera *Triodia*, *Plechtrachne*, *Symplectrodia* and *Monodia* (Jacobs 1992). They occur on rocks, sand and laterite (Beard 1981), usually with bare ground between the plants.

Hummock grasses may be too moist to burn (as in many parts of Rudall River National Park in the winter of 1992) or remain moribund and highly flammable for years (as in the Gibson Desert from 1988-1991); both the cured and green condition may exist in the same region at the same time (as at Rudall River National Park in 1992). The hummock grasslands of our study areas were largely of live material, the only apparently dead material being the immediate subcanopy leaves and the persistent remnants of leaf sheaths on the tillers. Hummocks at suitable moisture contents supported fire even without the growth of ephemerals in the usually bare areas between the plants. After 27 mm of rain overnight in January 1993 at Rudall River National Park, hummock grasses, straw coloured at first, became greener over the next few days (A.M.Gill, P.H.R.Moore and B.Ward, personal communication); a similar observation has been made in the Gibson Desert (N.D.Burrows, personal communication).

The coverage of hummocks over the ground may vary widely from near 100 per cent in some drainage lines to near zero on some rocky hills. Plant size, and shape (see above), may also vary widely. Plants 25-30 cm tall were common in the study area. Hummocks varied widely in density (i.e. number of plants per unit area), degree of aggregation (genets per hummock), aggregate shape and height, and within-plant bulk density (plant weight per unit volume); these attributes, along with moisture content and windspeed, have a major bearing on whether or not a fire will spread.

The appropriate manner in which to record the sizes and patterning of spinifex hummocks for prediction of fire spread needs further research. The only method reported so far has been the wheel-point (point quadrat) method (Griffin and Allan 1984; Burrows *et al.* 1991) which records plant presence or absence every 1 m along a transect. This method gives an estimate of cover and an index of pattern based on the number of consecutive wheel-point contacts made on plants, or on bare ground. On the basis that the fire spreads when the gap between plants is breached by a flame of a suitable length developed from a hummock of an appropriate size (Bradstock and Gill 1993), we need to know the length of flame developed from any hummock and the distance that flame will have to breach to ignite another. Bradstock and Gill (1993) found flame length to be correlated with hummock height and diameter for circular hummocks. For irregularly shaped hummocks, we have been exploring the idea of randomly selecting a hummock, selecting a random compass direction, measuring the longest intercept across the hummock in the chosen compass direction and aligned with the nearest neighbour, measuring the shortest distance from the edge of the hummock to the nearest neighbour along the same compass orientation ('gap width'), and measuring hummock height, width and length. Our results for gap widths (Fig. 1) show various frequency distributions (for 50 measurements per site) with gaps up to 2 m or more on some occasions; in our three study sites at Rudall River National Park, the frequency of gaps less than 60 cm wide was 70 per cent, 66 per cent and 46 per cent. While this idea for fuel measurement in discrete hummock-grass fuels needs to be tested in the light of experimental fires, it has potential in that it directs attention to the obstacles in the way of fire spread in discontinuous fuels. We do not know what the consequences to fire spread of the different frequency distributions of gap and hummock sizes may be but these could be predicted if isosceles triangles can be used as templates for spread, the template shapes and sizes representing flame shapes and dimensions in plan view.

Prediction of hummock sizes, and the widths of gaps between them, as a function of time since fire is a challenge waiting to be met. Plant cover, and therefore fuel loading (Griffin and Allan 1984), is a function of accumulated rainfall since the previous fire (Griffin 1992) but the way the pattern changes with time needs to be investigated.

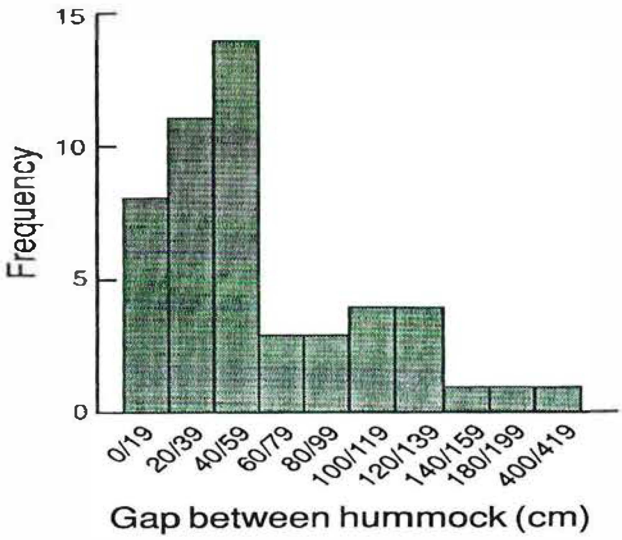

Figure 1. Frequency distribution of gap widths in a spinifex grassland fuel at the Rudall River site.

FIRE WEATHER

Fire weather indices provide a guide as to the effect of weather on fire behaviour. McArthur's (1967) 'fire danger meter' for eucalypt forests, asserts that the weather index given by the meter is 'directly related to the chances of a fire starting, its rate of spread, intensity and difficulty of suppression ...'. The index is based on windspeed together with factors contributing to fuel moisture. The index reflects the effects of weather on fire behaviour. The equivalent McArthur (1966) index for grasslands uses a different formula (Noble *et al.* 1980).

Relating weather data to the moisture content of fuels like *Triodia* - which may be green and live, or grey and dead - is difficult. Dead fuel moisture may be estimated from equations such as those of Byram and Jemison (1943) but a number of steps are involved (Fig. 2). Predicting live-fuel moisture may be even more difficult and involve inputs of soil moisture and vapour pressure deficit of the air (Fig. 2). However, the nature of moisture content changes with desiccation of the environment in these plants is unknown. Stems of hummock grasses (*ca* 2 mm diameter) were a significant proportion of the fuel that may be moist while leaves were drier. Here, we present an overview only, of the weather in the vicinity of Rudall River National Park.

The climate of the region may be characterized (using unpublished Bureau of Meteorology data from Telfer, 70 km to the north) as one with a low rainfall most of which falls in summer, high temperatures (*average* daily maxima in summer are near 40°C), low humidities (*average* relative humidity at 3.00 p.m. at Telfer for October is 11 per cent) and high evaporation rates. Telfer had an average annual rainfall of 291 mm over the period 1974-1991 with a marked summer incidence. The maximum average monthly rainfall was 84 mm in February. Average evaporation rate peaked in November at 15.0 mm day^{-1} and fell to a minimum in June when the value was 5.6 mm day^{-1}. Our experience was that, in summer, rainfall tended to occur as local storms so that the weather station record is not always a true reflection of the weather over the region.

Wind is a most important variable in determining whether or not a fire will spread in discrete fuels and, if the fire does spread, how fast it will travel. Wind may also affect temperatures of dead fuels (Byram and Jemison 1943) and of live fuels through transpiration. Average monthly 3.00 p.m. windspeeds at Telfer varied from 14-17 km h^{-1} with little variation throughout the year. Frequencies of winds were greatest in the 0-10 km h^{-1} class, the frequency then declining with each higher windspeed class. Wind speed within any one month may range up to about 60 km h^{-1}. In January 1993, at Rudall River National Park, we found that strong winds were associated with the presence of nearby storms (dry in our case) which were shortlived. Longer lasting cyclonic winds may be important at some times in some years.

To summarize: fire weather in the Rudall River area can involve frequent high temperatures, low humidities and low soil moistures, and occasional high winds.

THREE-STAGE FIRE MODELLING

Fire managers sometimes talk of fuels which either carry fire extremely well or not at all - 'stop-go' behaviour; such fuels are usually discrete. In discrete fuels, even dry ones, fires will not spread unless the wind is relatively strong. Fires in similar, but continuous, fuels will spread even without wind. When the wind is above the threshold for spread, the fire may spread at the same rate as in a continuous fuel. If so, the apparent 'stop-go' behaviour is explained (Fig. 3); the more discrete the fuel, the sharper the contrast in the 'stop-go' behaviour.

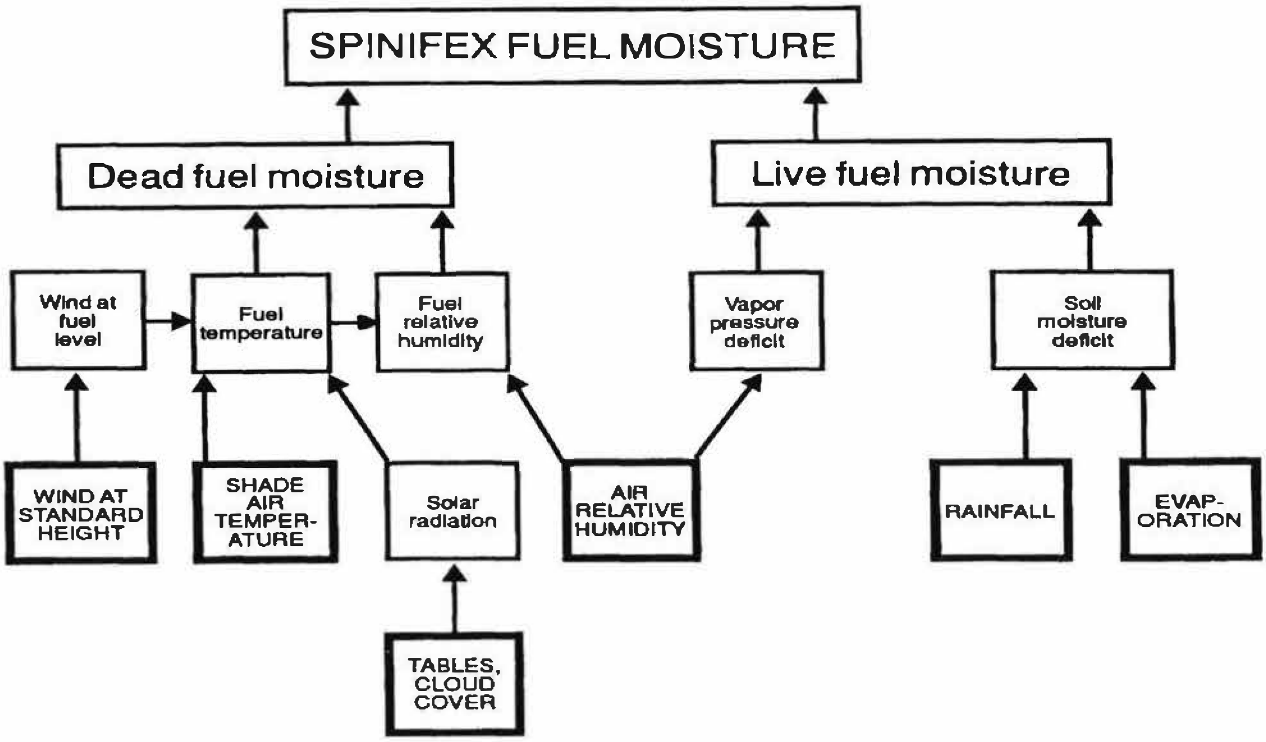

Figure 2. Simplified chart suggesting variables that may need to be considered for the prediction of fuel moisture from standard weather variables in order to predict sunlit dead-fuel (after Byram and Jemison 1943) and live-fuel moisture. Variables in double boxes are input variables.

For fires to spread in discrete hummock grass fuels, flames generated in the hummocks need to be long enough, winds need to be strong enough to angle the flames enough, gaps need to be narrow enough for the flames to breach them, and ignition delay time needs to be short enough for the 'receiver' hummock to ignite. We can imagine a number of factors limiting spread despite individual hummocks being ignitable. For example: if winds are too light, flame tilt may be inadequate for spread; with strong winds, flames may be horizontal, but the fire will not spread to the next hummock unless the reach of the flame at least equals the gap size; and, the fire will not spread if the flames do not impinge long enough on the hummock to cause ignition. To predict when fires would <u>start</u> to spread, therefore, requires that certain (multiple) threshold conditions need to be met. The same could be argued for continuous fuels although rate-of-spread models for continuous fuels are not usually designed to predict threshold conditions of spread - or when fires will go out. For example, the equations for the rate-of-spread meter of McArthur for forests (Noble *et al.* 1980) never zero in relation to moisture content (through its surrogates, air temperature and relative humidity), and, mathematically at least, a moisture content too high for combustion can be compensated for by a strong wind. In Rothermel's (1972) formulation, the problem is handled, in part only (Wilson 1985), by the use of a minimum moisture content for fire extinction.

Following Gill and Bradstock (personal communication), it is suggested that the scientific use of rate-of-spread models be *formalized* into three stages:

(i) a domain analysis to determine whether or not the data to be put into the rate of spread equations are within the limits of those used in equation formulation;

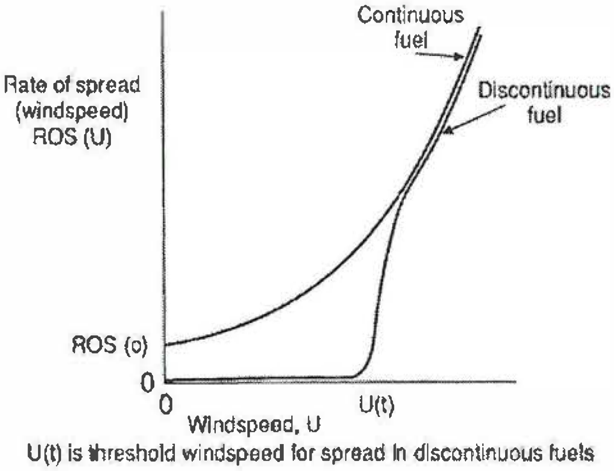

Figure 3. Graph illustrating the likely relationship between fire spread in continuous versus discrete fuels. In discrete fuels, fires will only spread after certain thresholds have been overcome; when this happens, fire spread may be quite rapid.

(ii) an initiation-of-spread analysis to predict the probability of spread, which, in its simplest form, would predict whether or not the fire would spread at all; and,

(iii) a rate-of-spread prediction.

For hummock grasslands, Griffin and Allan (1984) have given the domains to which their equations for predicting rates of spread apply; the conditions include a maximum windspeed of 3 m s^{-1} (*c.* 11 km h^{-1}) at 2 m height, air temperatures of 23 to 35°C and relative humidities from 12 to 40 per cent. For the Rudall River area, even average weather conditions are often outside these domains. In experimental Gibson Desert fires, the environmental variables under which fires were measured spanned a much wider range than those used in Central Australia by Griffin and Allan (1984), *viz.* windspeeds to 36 km h^{-1}, air temperatures to 50°C, and relative humidities from 14 to 48 per cent (Burrows *et al.* 1991), yet the same formulations used by Griffin and Allan could be used to predict fire rates of spread. Thus, Burrows *et al.* (1991) effectively expanded the domains of Griffin and Allan's (1984) equations as far as weather was concerned.

For fuel, a simplistic example shows the importance of keeping within domains. If Griffin and Allan's (1984) 'fuel factor' (inversely proportional to the square root of fuel moisture content) is extended beyond stated domains to include zero moisture content, its value becomes infinite, as does the predicted fire rate of spread.

Threshold conditions for spread have rarely been stated for any fuel type but various values have been suggested for windspeed in hummock grasses, *viz:*

- 16-24 km h^{-1} (McArthur 1972);
- 12-17 km h^{-1} (Burrows *et al.* 1991); and,
- <11 km h^{-1} (Griffin and Allan 1984).

At Rudall River in January 1993, it was found that winds gusting up to 50 km h^{-1} at 2 m height were insufficient to carry flames between hummocks, presumably because fuels were too moist, at 24 per cent, and flames too short to bridge the gaps between the hummocks (A.M.Gill, P.H.R.Moore and B.Ward, personal observation). All limits - gap width, wind strength, flame length, discrete-fuel loading, fuel moisture - not just any one, need to be overcome if fires are to spread.

The third stage in the prediction of the rate of spread of the fire is the use of models, usually in the form of mathematical equations. Griffin and Allan's (1984) model was the first for fires in spinifex grasslands. For fires in the same type of grassland, but in the Gibson Desert, Burrows *et al.* (1991) found that spread rates, using the same formulae as Griffin and Allan (1984), were less than half those predicted. The main reason for this wide discrepancy may involve the way in which fuels were evaluated (see above). Similarly, the successful use of the square of windspeed to predict fire spread in the Gibson Desert (Burrows *et al.* 1991) may not have general application; some modification of equations would appear to be necessary for their application to areas with different fuel conditions.

CONCLUSION

Despite the considerable advances made in our knowledge of fires in spinifex grasslands in the last decade we still have much to learn if we are going to be able to predict when discrete hummock-grass fuels will ignite, whether or not fires in hummocks will spread, and how fast they will travel once they do spread. We need to know: how to predict the moisture contents of hummock grasses; how to characterize and predict fuel distributions as a function of time since fire or cumulative rainfall since fire; how to depict fire weather in arid lands; and how to parameterize the factors limiting fire spread in hummock-grass fuels.

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Predicting fire spread in Western Australian mallee-heath

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ABSTRACT

The ability to reliably predict whether a fire will sustain itself, and what its resultant rate of spread and intensity will be are key elements in the safe and effective application of prescribed fire in mallee-heath shrublands in Western Australia. A program of experimental burning has been conducted in *Eucalyptus tetragona* mallee-heath at the Stirling Range National Park to identify factors critical to the sustained spread of fire, and to determine the effect of burning conditions on the forward spread rate of fires ignited from a 200 m line source. Fires ignited when the moisture content of the shallow litter beneath the shrub layer was more than 8 per cent (of oven dry weight) typically failed to sustain a continuous flame front and did not spread extensively. However, when the shallow litter layer was below 8 per cent moisture content fire fronts sustained and spread regardless of wind speed. Other indices of fuel dryness, and simple weather variables including air temperature and relative humidity did not satisfactorily discriminate between the fires that spread and those that did not.

Forward spread rates of fires burning in fuels below the critical moisture content threshold was strongly related to wind speed, and spread rates of up to 2500 m h^{-1} were recorded when wind speeds (at 10 m height in the open) were around 25 km h^{-1}. Forward spread rates of experimental fires correlated poorly with the Grassland Fire Danger Index from the McArthur Mark 4 meter which forms the basis for fire danger rating in rural areas of Western Australia. Models developed to predict fire behaviour in chaparral shrublands in the United States were of little value for determining whether or not a fire would spread, and consistently under-predicted forward spread rates in mallee-heath.

In mallee-heath it seems that the probability of a fire sustaining must be determined as a preliminary step to predicting the forward rate of spread; the former factor is moisture-dependent while the latter is wind-dependent. To predict fire behaviour in the field situation simple but reliable methods for estimating the moisture content of the shallow litter layer in mallee-heath are required.

INTRODUCTION

Shrublands are an important and widespread fuel type in southern Western Australia, and despite extensive clearing for agriculture still occupy an area of some 5.5 million hectares (Beard 1984). Mallee-heath is the most common form of shrubland on the southern sandplain which extends between Albany (35° S 118° E) and Israelite Bay (34° S 124° E). Mallee-heath communities are characterized by a stratum of low shrubs up to 1 m or so tall with a scattered overstorey of short, multi-stemmed eucalypts up to 4 m tall. A wide variety of plant species occur in these diverse communities, with *Eucalyptus tetragona* being one of the most common and widespread of the mallee eucalypts present. Extensive tracts of mallee-heath occur on lands managed by the Western Australian Department of Conservation and Land Management (CALM), including the Stirling Range, Fitzgerald River and Cape Arid National Parks, and on adjacent areas of vacant Crown land. Much of this area is relatively remote from population centres and management resources, and often inaccessible because of difficult terrain and limited roading.

Like most of southern Western Australia the south coast is prone to periodic fires. Severe fire weather conditions characterized by extreme high temperatures, low humidity and strong winds occur regularly each summer, particularly in association with the formation of pre-frontal low pressure troughs (Hanstrum *et al.* 1991). Trough movement is commonly accompanied by dry lightning storms which may ignite fires over a widespread area. Air photographs consistently show evidence of extensive fires over the past 50 years, even in areas remote from land clearing and other potential sources of human-caused ignition. More recently, three fires started by lightning burnt over 100 000 ha of the

Fitzgerald River National Park on a single day in December 1989 (McCaw *et al.* 1992). Further lightning-caused fires burnt some 750 000 ha of CALM estate and vacant Crown land in the Esperance-Norseman area over the summer of 1990-91 (McCaw 1992).

CALM undertakes fire management on lands under its control with the objectives of protecting life and property from damage by wildfire, and maintaining and enhancing environmental values. This latter objective may in some cases necessitate the temporary protection of an area from fire, while in other cases prescribed fire may be used deliberately to regenerate plant communities (McCaw and Gillen 1993). Prior to about 1985 the most common fire management strategy for large reserves in the south coast region was to undertake limited fuel reduction burning within narrow buffer strips located at the interface with neighbouring lands of different tenure. This approach has proved problematic in mallee-heath where slight changes in fuel and weather conditions can lead to dramatic alteration in fire behaviour. If burning conditions are sub-optimal fires do not sustain, leading to inefficient use of resources and ineffective fuel reduction within buffer strips. At the other end of the scale fires which exceed prescribed limits for fire intensity may escape from narrow buffer strips, thus posing a threat to neighbouring lands and potentially endangering personnel at the fireface.

In recent years the feasibility of using aerial ignition to burn strategic strips and mosaics within large unroaded blocks has been investigated and found to have considerable promise as a fire management technique in mallee-heath communities. Successful implementation of this technique depends on a sound understanding of fire behaviour and, in particular, the ability to reliably predict the onset and cessation of sustained fire spread.

In 1989 CALM commenced a study of fire behaviour in mallee-heath, the principal objectives of which were to (1) identify factors critical to the sustained spread of fire, and (2) develop a fire behaviour guide for predicting rate of spread and fire intensity from weather and fuel variables readily measured in the field. The study was based around a program of experimental burning undertaken in the Stirling Range National Park, and involved personnel from CALM's Manjimup Research Centre and South Coast Region. Experimentally-derived data have been supplemented by information gained opportunistically during prescribed burning operations and wildfires. Preliminary results from the study were presented by McCaw (1991). The purpose of this paper is to present updated results from the experimental burning program, and to examine the spread of experimental fires in relation to the Grassland Fire Danger Index from McArthur's Mark 4 meter (1973), and fire spread predictions from two models developed for chaparral shrublands in the United States.

METHODS

Experimental Burning

In March 1989 sixteen plots, each 4 ha in area, were demarcated in a uniform area of *E. tetragona* mallee-heath near Two Mile Lake (34° 31"S, 118° 13' E) on the southern boundary of the Stirling Range National Park. The area experiences a mediterranean climate with an annual rainfall of about 470 mm. Plots were located on flat terrain at an elevation of 160 m above sea level in an area that had last been burnt in 1969 by a summer wildfire. Plots were grouped into cells of four, each of which was enclosed within a 100 m wide buffer strip created by scrub-rolling and burning the vegetation; the system of buffer strips permitted experimental fires to be conducted under relatively severe burning conditions without undue risk of escape. Two larger plots (16-50 ha in area) established nearby were also burnt as part of the experimental program.

Mallee eucalypts common throughout the area included *E. tetragona*, *E. pachyloma*, and *E. marginata*. The shrub layer was species rich with between 50 and 60 species of vascular plants present in a 10 m x 10 m quadrat. Dominant shrubs included species of *Hakea*, *Dryandra*, *Isopogon* and *Banksia*. Fuels were assessed on thirty 10 m x 1 m transects distributed across the site which were harvested in four height classes and sorted by size classes into live and dead fractions (for a more detailed account of this procedure see McCaw 1991). The mean fuel load for the site was 12.4 t ha^{-1}, consisting of 4.5 t ha^{-1} of litter, 3.1 t ha^{-1} of dead shrub components (<25 mm diameter) and 4.8 t ha^{-1} of live shrub foliage and twigs (<6 mm diameter). The layer of litter on the ground was light and discontinuous, with a projected cover of about 75 per cent.

Weather data recorded during each experimental fire included air temperature, relative humidity, 10 m open wind speed, and global solar radiation. Wind speed was measured using a Unidata anemometer mounted on a 10 m tall Clark mast located 50 m upwind from the side of the plot which was to be ignited. At the time of each fire five replicate samples of four fuel fractions were collected for determination of oven dry moisture content, as follows:

- shallow litter (<10 mm deep) from beneath shrubs,
- deep litter (10-30 mm deep) from beneath mallee clumps,
- elevated dead fuel (<6 mm diameter) from shrubs,
- live foliage from *Dryandra pteridifolia*, a species of low shrub common throughout the study area.

Plots were ignited on the upwind side using a vehicle-mounted flame thrower to light a 200 m long line of fire; ignition was generally completed within a two-minute period. Fires were allowed to spread across the plot with the wind and were contained by the scrub-rolled buffer strips.

Fire spread was measured using buried electronic timers activated by the passage of the flame front, with elapsed times calculated in relation to a master clock started at the commencement of ignition. Reliable timer data were used to prepare fire spread contour diagrams for each plot from which the forward rate of spread of the fire was determined. Spread contours were checked for consistency with the prevailing wind direction during each fire, and compared with low-oblique photographs and visual observation recorded during each burn. Inconsistencies between observed fire behaviour and fire spread contours plotted from timer data resulted in three plots being excluded from some stages of the analysis.

Comparison with Predictions from Existing Fire Spread Models

The forward rates of spread of fourteen experimental fires were compared with predictions made using three existing fire spread models: the Mark 4 Grassland Fire Danger Meter developed by McArthur (1973); the Rothermel (1972) fire spread model; and the Arizona oak chaparral model of Lindenmuth and Davis (1973). Three fires were excluded from this analysis because forward spread rates could not be reliably determined, and a further fire was excluded because it was in a fuel type that was not representative of the fuels in the remainder of the study area.

Although not developed for predicting fire spread in shrubland fuel types the McArthur Grassland meter is used for fire danger rating in rural areas of Western Australia, including setting of restrictions on prescribed burning. The extent to which calculated fire danger indices reflect the severity of fire behaviour in mallee-heath is therefore an important issue for fire authorities and bush fire brigades working in this fuel type. The Grassland Fire Danger (GFD) index at the time of each experimental fire was calculated from the Mark 4 meter using measured air temperature, relative humidity, 10 m open wind speed and a constant curing factor of 100 per cent. This level of curing was chosen because most large wildfires in mallee-heath have occurred in summer and autumn when pastures and crops were fully cured.

Rates of spread were predicted from the Rothermel model for fuel model 6 using the nomograms provided in Rothermel (1983). This fuel model was selected from Anderson (1982) on the basis of its similar appearance and fuel loading characteristics to mallee-heath.

Variables used to calculate the rate of spread with the Lindenmuth and Davis oak chaparral model were measured air temperature, relative humidity, wind speed, and solar radiation. Foliar moisture content was held constant at 85 per cent which was considered to be a representative value for mature foliage of a range of mallee eucalypts and shrubs (McCaw, unpublished data). No correction was made for chemical coefficient (Davis and Dieterich 1976) as the influence of this factor on fire behaviour in mallee-heath is unknown.

Predictions from the two United States fire models are based on wind speeds measured at 20 feet (6.1 m approximately) above the ground. Wind speeds measured at 10 m were adjusted to 6.1 m equivalents using the Applied Meteoroligical Tables of Beer (1990) assuming a roughness length of 0.2 m. In the case of the Rothermel model a standard correction factor of 0.4 was then applied to adjust wind speeds to mid-flame height, as specified for exposed fuel situations by Rothermel (1983, p.33).

RESULTS

Factors Critical to the Sustained Spread of Fire

Eighteen experimental fires were ignited over a wide range of burning conditions ranging from cool moist weather in spring and late autumn to hot, dry weather in summer (Table 1).

Data from Plot 3 were excluded at the outset from analysis because this plot contained very sparse fuels that were not representative of the remainder of the site. Most of this plot failed to burn despite the burning conditions at the time of ignition being relatively severe, with an air temperature of 32°C, relative humidity of 40 per cent, wind speed of 17 km h^{-1}, and shallow litter moisture content of 7 per cent.

TABLE 1

Summary of weather conditions and fuel moisture during 18 experimental fires in mallee-heath.

Moisture content (MC) data are presented for shallow (SHALLOW) and deep (DEEP) layers of litter fuel, dead elevated fine fuel (DEAD), and live *Dryandra* foliage (LIVE). Weather data are mean values recorded over the duration of experimental fires which was generally less than ten minutes, or over a default period of ten minutes in cases where fires failed to spread.

VARIABLE	UNIT	MEAN	RANGE
Air temperature	°C	25	20 - 36
Relative humidity	%	44	14 - 63
Wind speed	km h^{-1}	16.6	5 - 25
Solar radiation	W m^{-2}	606	271 - 912
$MC_{SHALLOW}$	%	6.6	3 - 14
MC_{DEEP}	%	10.0	4 - 32
MC_{DEAD}	%	9.9	6 - 16
MC_{LIVE}	%	82	68 - 90

Five of the experimental fires failed to spread following ignition. The characteristic feature of these fires was that the flame front became discontinuous within a short time following ignition (i.e. within one or two minutes) and did not spread any significant distance from the ignition line. Flames only persisted in favourable fuel situations such as the continuous litter beds beneath mallee clumps and patches of low shrubs with a substantial component of elevated dead fuel.

In contrast, a second group of 12 experimental fires was characterized by continuous flame fronts which persisted even in areas where fuels were light and patchy. These fires spread freely across the plot according to the strength and direction of the wind, and in some cases developed rapidly into a crown fire within a minute or so of ignition. The distinction between these and the former group of failed ignitions was clear and individual fires were assigned to one or other category without difficulty.

Exploratory analysis of the factors critical to sustained fire spread was undertaken using scattergrams (Fig. 1) with a range of weather and fuel moisture variables plotted against wind speed. Of the weather variables, air temperature and relative humidity did not clearly discriminate the fires that spread from those that didn't. The probability of a fire sustaining did, however, appear related to solar radiation as no fires sustained when radiation was reduced below 400 W m^{-2} by heavy cloud cover.

Fires which did not spread following ignition were consistently associated with higher moisture contents for each of the dead fuel fractions sampled (Fig. 1). The variable which most clearly distinguished the fires that spread from those that did not was the moisture content of the shallow litter layer. In all cases fires spread when the moisture content of the shallow litter was below 8 per cent. This effect was independent of wind speed across the range of the experimental burning conditions which included wind speeds from 5 - 25 km h^{-1}.

Forward Rate of Spread

Forward rates of spread were determined for nine of the experimental fires with three fires being excluded because timer data were unreliable, or because shifts in wind direction during the course of the fire made it difficult to define a true fire front. One fire conducted under very light winds (<5 km h^{-1} at 10 m) was retained in the analysis despite considerable variability in wind direction. This fire did develop a recognizable front, although the direction of spread was not entirely consistent with the mean wind direction over the course of the fire. In this case the spread of the fire front was determined more by occasional strong gusts than by the average wind condition.

The effect of weather and fuel variables on forward rate of spread was tested for the nine fires remaining in the data set using multiple linear regression, with both raw and transformed data used to determine the equation of best fit. Wind speed was the only variable to have a significant effect on forward spread rate, with the relationship exhibiting a linear form over the range of the experimental data (Fig. 2). The regression equation developed for untransformed data accounted for 59 per cent of variation in rate of spread -

$$FROS = 1.51 + 90.6\ WIND \quad R^2 = 0.59,\ P<0.05$$

where *FROS* is forward rate of spread in m h^{-1} and *WIND* is wind speed in km h^{-1} measured at 10 m height in the open. The tendency for variance to increase with increasing rate of spread was rectified by log transforming rate of spread data, which resulted in an equation of slightly improved fit -

$$lnFROS = 5.64 + 0.09\ WIND \quad R^2 = 0.66,\ P<0.01$$

Comparison with Predictions from Existing Fire Spread Models

Experimental fires were conducted at GFD indices ranging from 3 (Low) to 23 (Very High); for all but one of the fires the index was below 10. Indices associated with the group of fires that failed to spread following ignition spanned a similar range (3-9) to those associated with the group of fires that did initiate and spread (Fig. 3a). The three fastest spreading experimental fires, which had forward spread rates exceeding 2000 m h^{-1}, were associated with GFD indices of 8 or above. However, one of the fires that failed to sustain was ignited at an index of 9. These results suggest that the GFD index is unlikely to prove useful as an indicator of whether or not a fire will initiate and spread. Also, there is unlikely to be a consistent relationship between the GFD index and the rate of forward spread of mallee-heath fires, at least for burning conditions resulting in an index less than 10.

Rates of forward spread predicted for fuel model 6 using the Rothermel fire spread model ranged between 80 m h^{-1} and 760 m h^{-1} (Fig. 3b). Fires that failed to initiate and spread were not consistently associated with low predicted rates of spread. Forward spread rates were positively correlated with predicted spread rates for fuel model 6 but were typically three to four times faster than predicted by the model. For the subset of nine fires that did initiate and spread, predicted and observed spread rates were related by the following linear regression equation:

$$FROS_{OBS} = 94 + 2.97\ FROS_{PRED} \quad R^2 = 0.59,\ P<0.05$$

where $FROS_{OBS}$ is the observed rate of spread in m h^{-1}, and $FROS_{PRED}$ is the predicted rate of spread for fuel model 6 in m h^{-1}.

The oak-chaparral model predicted forward spread rates ranging from 177 m h^{-1} to 388 m h^{-1} for the fourteen experimental fires and, as with the Rothermel model, there was no consistent association between low predicted rates of spread and failed ignitions (Fig. 3c). Predicted rates of spread bore no relation to, and were consistently much lower than those observed in mallee-heath.

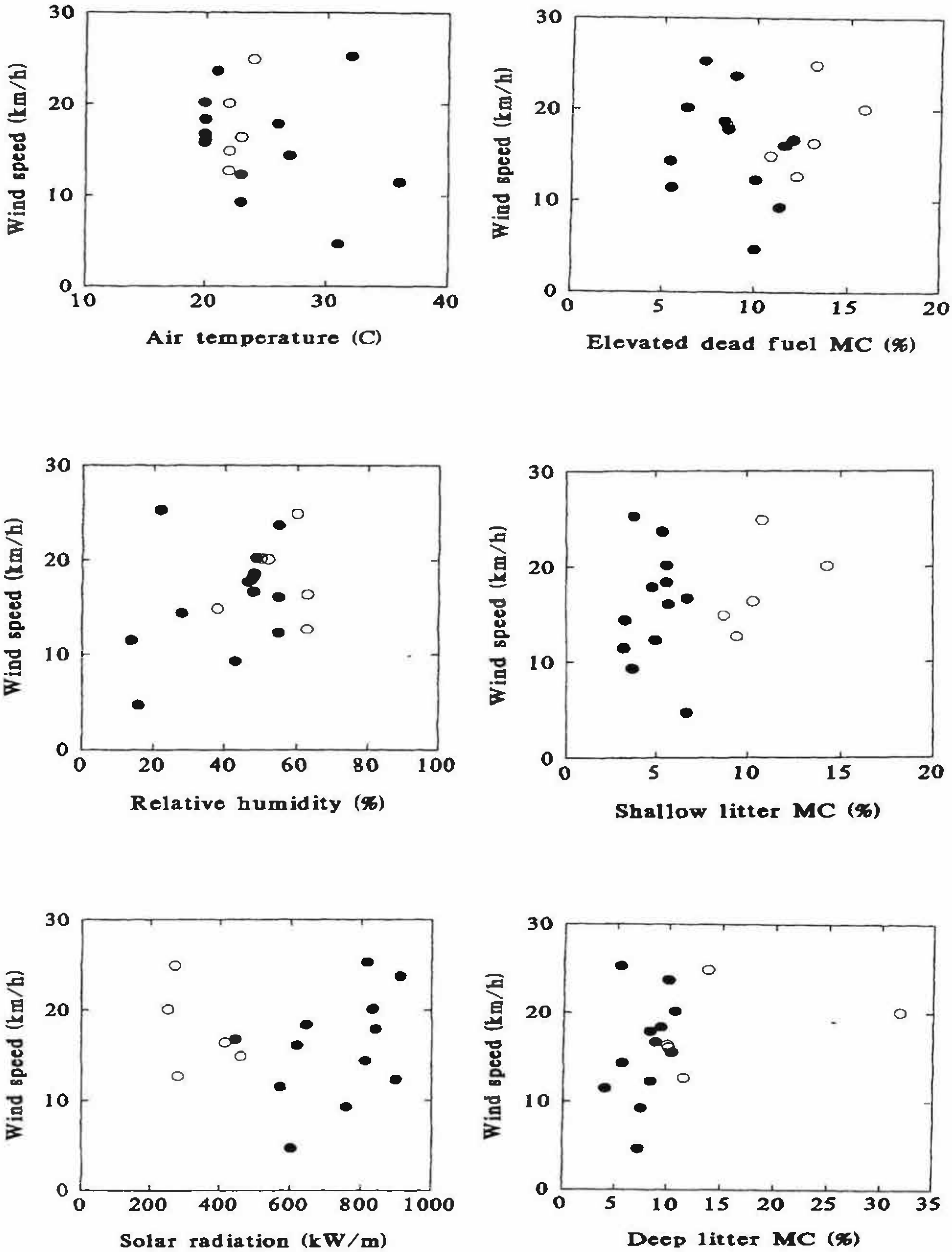

Figure 1. Scatterplots depicting environmental and fuel moisture variables for 17 experimental fires in mallee-heath. Fires represented by hollow symbols failed to spread following ignition; fires represented by shaded symbols spread freely following ignition.

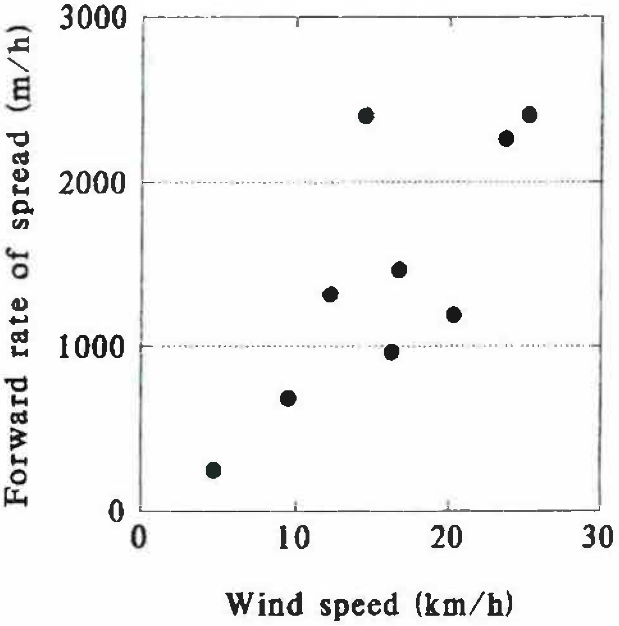

Figure 2. Forward rate of spread plotted in relation to 10 m open wind speed for 9 experimental fires for which reliable estimates of spread rate were available.

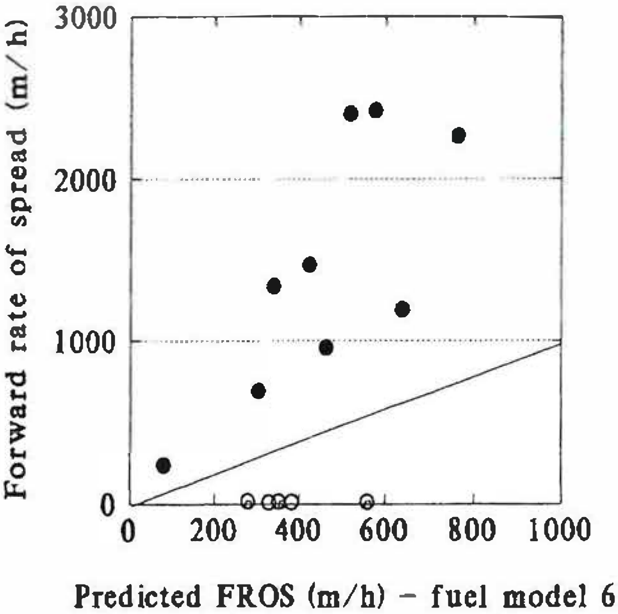

Figure 3b. Forward rate of spread of selected experimental fires plotted in relation to: rate of spread predicted for Fuel Model 6 using the Rothermel fire spread model.

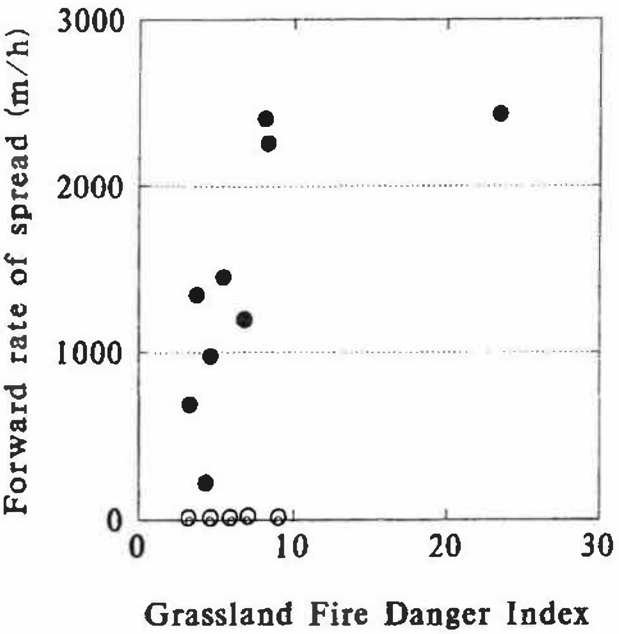

Figure 3a. Forward rate of spread of selected experimental fires plotted in relation to: the M^c^Arthur Mark 4 Grassland Fire Danger Index.

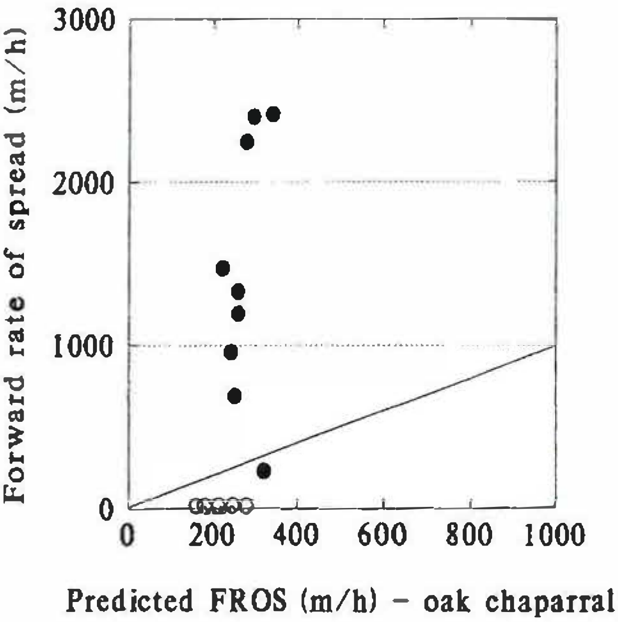

Figure 3c. Forward rate of spread of selected experimental fires plotted in relation to: rate of spread predicted by the Lindenmuth and Davis oak chaparral model.

DISCUSSION

The process of predicting fire spread in mallee-heath would seem to require two distinct steps: firstly, determining whether burning conditions are sufficient for a fire to initiate and spread and, if this is the case, then making a prediction of the forward rate of spread. Evidence from the experimental fires at the Stirling Range indicates that the probability of a fire initiating and spreading in mallee-heath is largely dependent on the level of fine fuel dryness, while wind speed has a dominant effect on the forward spread rate.

Sudden transitions in fire behaviour, apparently linked to slight changes in air temperature, relative humidity and wind speed have been reported to be a characteristic feature of fires in mallee-heath fuel types in southern Western Australia (McCaw *et al.* 1992). The existence of a distinct threshold level of moisture content required for continuous fire spread provides an explanation for this characteristic behaviour. Experimental data indicated that the critical level of fuel dryness required for fire to spread was a moisture content of 8 per cent in the shallow litter layer. Although no data are yet available for fires burning under conditions of strong wind (>20 km h^{-1}) with shallow litter in the 8-11 per cent moisture content range, the extent to which strong winds can offset the controlling effect of fuel moisture on fire spread appears to be quite limited in mallee-heath. Threshold moisture contents will almost certainly vary according to the characteristics of the fuel bed, with very dry conditions being necessary for fire spread in highly discontinuous fuels. This was clearly illustrated in the case of Plot 3 where the fire failed to spread following ignition despite a moisture content of 7 per cent in the shallow litter layer.

The observation that burning conditions must reach certain critical threshold before fires will spread continuously has been made for several other discontinuous fuel types including hummock grasslands in the Australian arid zone (Burrows *et al.* 1991; Gill *et al.* 1995), and oak-chaparral shrublands (Lindenmuth and Davis 1973) and pinyon-juniper woodland (Bruner and Klebenow 1979) in the south-western United States. Burrows *et al.* (1991) identified a critical range of wind speed between 12 and 17 km h^{-1} (measured at 2 m height above ground) necessary for fires to spread in spinifex (*Triodia basedowii* and *Plechtrachne schinzii*) hummock grasslands in the Gibson Desert of Western Australia, and reported that this threshold was unaffected by fuel moisture content up to a level of 30 per cent moisture content. The dominant effect of wind on fire spread in hummock grasslands has been attributed to the horizontal discontinuity of the fuel bed which consists of discrete hummocks separated by expanses of bare ground. For fires to spread in such discrete fuels the wind must be sufficiently strong to tilt flames across to the next hummock for long enough to ignite it (Gill *et al.* 1995). Threshold wind speeds for continuous fire spread are therefore likely to vary according to the size, shape, condition and spatial distribution of hummocks (Bradstock and Gill 1993). In the case of oak-chaparral shrubland Lindenmuth and Davis (1973) noted that 'people experienced in Arizona chaparral have always maintained that chaparral either burns fiercely or does not burn at all - no gradation in between'. Experimental burning studies by Lindenmuth and Davis confirmed this rule of thumb and identified a critical rate of spread of about 360 m h^{-1}; burning conditions had to be sufficient to generate spread at or above that level before fires would spread across country. Wind was found to be a limiting factor with speeds of around 13 to 15 km h^{-1} (at 10 m height) needed for fires to spread satisfactorily, provided temperature and fuel moisture conditions were favourable. Based on the descriptive account of fuel characteristics in their paper and my own field observations of oak-chaparral it seems that individual oak clumps are relatively discrete and that this fuel type is more discontinous than mallee-heath, probably being intermediate between mallee-heath and hummock grassland. For pinyon-juniper woodland Bruner and Klebenow (1979) found that the likelihood of a fire sustaining could be predicted from a simple score comprising the wind speed, air temperature and percentage vegetation cover; unfortunately the height at which wind speeds were measured was not specified in their paper. While having some application in localized situations this approach has the limitations of not identifying the relative importance of the various factors contributing to the score, and not indicating potential forward rate of spread, and hence fireline intensity.

The Rothermel model showed greater promise than either the McArthur grassland or the oak chaparral model for predicting rate of spread in mallee-heath, although none of the three models tested adequately identified the conditions under which fires would not spread. There may be scope for improving predictions from the Rothermel model by developing a specific fuel model (Burgan and Rothermel 1984) for mallee-heath, and by using an improved technique to estimate mid-flame wind speed (Durre and Beer 1989). The absence of a clear relationship between observed fire behaviour in mallee-heath and the predictions of the oak chaparral model are probably explained by the strictly empirical origins of this model which make it unsuitable for broader application. Differences between mallee-heath and oak chaparral in the relative flammability of live foliage, and in the effects of moisture content and chemical composition on live foliage flammability may be important in this regard.

To predict fire behaviour in the field situation simple but reliable methods for estimating the moisture content of the shallow litter layer in mallee-heath are required. A range of other models for estimating fuel moisture content are available (Viney 1991) and the application of these models in mallee-heath should be evaluated. Direct measurement of fuel moisture content in the field with suitably calibrated grain moisture meters or similar devices provides an alternative method. However, meters currently in use, such as the Marconi, are unsuitable for measuring leaf litter fuels drier than about 10 per cent moisture content and would therefore have limited application in mallee-heath where there is a requirement to accurately measure the moisture content of very dry fuels. New technologies for measuring fuel moisture content in the field should therefore also be investigated as the opportunity arises.

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Fire from a flora, fauna and soil perspective: sensible heat measurement

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ABSTRACT

The two main areas of interest in landscape fires are the protection of human life and property, and the ecological effects of fire regimes. The protection of human life and property involves the provision of adequate fire prevention and suppression strategies and requires an understanding of aspects of fire behaviour such as rate of spread, flame height and fire intensity. However, an understanding of the ecological effects of fire is as complex as the different ecological systems themselves. In this paper I address the issue of characterizing fires in a way that will help forest managers to predict the likely effects of a fire on a forest community under different environmental conditions.

Three methods of measuring the sensible heat output from different fires, the billy calorimeter, the heat sensitive plate and scorch height are presented. Sensible heat is defined as the integration of heat in the forms of radiation, convection and conduction resulting from a fire and the environmental conditions at the time of the fire. The fire behaviour and environmental factors affecting the sensible heat flux measured are explored. Measures of sensible heat are used to correlate the effects of fire on tree boles, soil chemistry and soil invertebrates. The usefulness of these measures as measures of the ecological effects of fire are discussed.

It is concluded that sensible heat output from a fire is a key criterion for determining the effect of individual fires on flora, fauna and soils.

INTRODUCTION

The need to control and use fire in natural and cultivated landscapes has led to an interest in describing fires and the conditions under which they occur. However, as Van Wagner (1965) pointed out, the subjective nature of fire descriptions made it nearly impossible to compare different fires in different places and at different times described by different people. Byram (1959) significantly advanced fire research by proposing a measure of fire intensity which combined the heat of combustion of the fuel burnt, the amount of fuel burnt and the rate of spread of the fire front. At last here was an objective measure of fire behaviour which described the rate of energy output along a given length of the fire front. Over the past 30 years Byram's fireline intensity has been used as an essential descriptor of any fire being studied for research purposes, but is less commonly used by fire managers or fire fighters. However, by itself, Byram's fireline intensity does not give a complete picture of fire behaviour from a fire controllers point of view nor a fire ecologists point of view (Tangren 1976).

Fire controllers are interested in aspects of fire behaviour such as its intensity, its rate of spread, spotting distance and flame height, and while fire ecologists are also interested in fire intensity, they are also interested in the total amount of heat output, the duration of the heating, the moisture conditions of the soil and vegetation (alive and dead) and the general fire regime. McArthur and Cheney (1966) suggested that fire intensity, duration of heating, and the proportion of fuel available at the time of the fire were important fire characteristics needed to adequately describe a fire for ecological studies. While this may be true, nearly 30 years later, our ability to relate the effects of fire on flora, fauna and soils is coarse and crude. This is firstly due to the complexity of the interactions between the physiological state of plants and animals at the time of a fire, with the fire, and secondly the variability of fire, soil, plants and animals within even a small area of a bushfire. For this reason this paper will focus on some measures of sensible heat output from bushfires which integrate many of the variables involved in the ecological effects of fire.

The idea of a sensible heat measuring device in the flaming zone of the fire was first reported by Martin (1963). Martin used roasting dishes set at one foot (0.3 m) and four feet (1.2 m) above ground level to measure the heat load of a forest fire. No results from these calorimeters were recorded. Later Beaufait

(1966) used one gallon (4.5 L) paint tins filled with 3 L of water, painted black on the outside and fitted with a lid with a 1 cm hole in the middle, to measure the amount of heat transferred to the calorimeter by measuring the amount of water lost through evaporation and the rise in temperature of the water remaining in the calorimeter. Beaufait reported that there was a wide variation in response to different fires, but did not report on how the output was being used. Knight (1981) described a development of another calorimeter, the billy calorimeter, which used a sensing device to record the duration and rate of heating (heat flux) as well as the total heat load. However, Knight found that the main factor affecting trees and vegetation were related to the increase in temperature of the billy rather than the rate of increase. As with Beaufait's billy, Knight found that the billy temperature was related to the amount of fuel burnt (and hence the total heat output) rather than the fire intensity.

Heat-sensitive materials have been used to measure the sensible heat below the flaming zone. These materials undergo recognizable and irreversible changes and offer a cheaper and more convenient alternative to the use of thermocouples (e.g. Raison *et al.* 1986) or other electrical sensing devices (e.g. Hodgkinson *et al.* 1982). In addition, heat sensitive materials are more easily placed in many locations within the one fire and can give a good spatial measure of temperature variability (Hobbs *et al.* 1984).

Some of the first quantitative work on the impacts of fire was done by Beadle in 1940. Beadle used organic compounds with different melting points in glass tubes to determine the maximum temperature reached by the soil subjected to a surface fire. He measured the temperature profile in the soil by placing these tubes at various depths. From the results of this work and a knowledge of the heating effect on lignotubers and seeds, he was able to predict the effects of different fires on the regeneration process.

Fire ecologists are usually interested in the time/temperature profile of the soil during and after a fire. A number of well known changes take place depending on the temperature reached by the soil and these are summarized in Walker *et al.* (1986). Temperatures between ambient and 125°C usually affect biological activity in the soil, between 200°C and 600°C affect the soil chemistry and temperatures greater than 600°C affect the physical nature of the soil. Usually, a moving fire only affects the top 1 to 3 cm of the soil (e.g. Tomkins *et al.* 1991; Raison *et al.* 1986), but a stationary fire such as a burning log pile can have an affect more than 30 cm deep (e.g. Tunstall *et al.* 1976; Cromer and Vines 1966).

Three methods of measuring sensible heat are discussed in this paper: the billy calorimeter, the heat sensitive plate and scorch height. Results are presented of measurements made in 35 low intensity fires (< 500 $kW.m^{-1}$) in mixed eucalypt forest in the foothills of north-central Victoria. Each of the three methods is discussed in turn.

METHODS

This work was conducted as part of the Fire Effects Research Program in the Wombat State Forest, Victoria. A detailed description of this study and its component projects can be found in Tolhurst and Flinn (1992).

Site Description

There are five study areas located on the Great Dividing Range within the Wombat State Forest which is 80 km north-west of Melbourne, Victoria (latitude 37°25', longitude 144°15'). The elevation of the study areas ranges between 550 m and 730 m. The average annual rainfall is 890 mm with at least 30 mm falling in each month of the year and about 70 per cent falling in winter and spring. Air temperatures in the region are cool, with average monthly temperatures being less than 6°C for three months of the year (June, July and August) and about 15°C in summer, although maximum temperatures may exceed 37°C in summer. The study areas are in forest comprising Messmate (***Eucalyptus obliqua*** L'Herit), Candlebark (***Eucalyptus rubida*** Deane & Maiden), and Narrow-leaf Peppermint (***Eucalyptus radiata*** Sieber ex DC) in the approximate ratio 5:3:2 respectively. The soils are yellow podzols derived from Ordovician sedimentary rocks.

Instrument Description

The Billy Calorimeter used in this study was based on the design of Knight (1981). The billy was a 3 L 'Milo' tin, painted with matt black paint on the outside and with a temperature sensing device inserted through the base of the billy. A lid was placed on the billy and water in the billy was kept stirred with a toy boat motor and propeller assembly powered by a single AA sized battery. The temperature sensing device in the billy was an analogue device (AD590) connected in series with a 10 k resistor and a 9 V transistor battery which gave a 10 mV change in output for each 1°C change in temperature. A cable was run underground from the billy to a safe area outside the plot where the changes in voltage (temperature) were manually recorded by reading the voltage across the resistor with a multimeter. The necessity of burying the cable limited the distance from the fire edge to the recorder to about 30 m. Later, after much trouble, an electronic circuit was designed to convert the voltage output from the billy to a frequency which was then sent via a frequency modulated (FM) transmitter to a radio receiver with another circuit which converted the frequency signal back to a voltage which was measured up to 150 m away from the fire's edge.

The heat sensitive plates were made from 0.3 mm thick aluminium flashing. The flashing was roughened with sandpaper on one side and a series of Thermochrom crayons (Faber Castell) were marked on the plate. The plate was then folded in two and hammered flat to ensure no blackening of the marked surface could occur in the fire. The performance of the crayons was tested in a muffle furnace and an interpretation chart was designed for estimating the temperature and duration of heating to which the plates were exposed (Appendix 1). The final plate measured approximately 5 cm x 12 cm. Earlier laboratory studies by the author had shown that the use of ceramic tiles as a carrier for the crayons significantly increased the time necessary for the a crayon to change colour, and when the tile was wrapped in aluminium foil as would be necessary for use in a fire situation, this time was increased further. Aluminium flashing has a lower heat capacity and better conducting properties than a ceramic tile and was found to satisfy our needs.

Treatments and Measurements

The results presented here are from experimental fires on 20 plots (4 at each of the 5 study areas) covering an area 35 m x 35 m. Each plot was within a larger fire area ranging in size from 3 ha to 35 ha (15 ha average). A 40 m line of fire was ignited downwind and/or downslope of the plots to achieve a headfire through the plot. Table 1 summarizes the fire behaviour and weather conditions. Some plots were burnt three times over a six-year period.

Fifteen fine fuel samples (< 6 mm) were collected before and after burning from 0.1 m^2 quadrats on each plot, oven dried at 105°C, and weighed to determine the fuel load and the amount of fuel burnt. Fire behaviour observations were made at 15 or more predetermined locations within each plot marked with a 1 m high post. Rate of spread was measured between each reference point, flame height was visually estimated and flame angle was measured from photographs taken as the fire front passed each point. Rate of spread, flame height, and flame angle were recorded at each of these points and then averaged for the plot. Fine fuel moisture contents were measured about every 30 minutes for the duration of the fire using a Marconi Moisture Meter (electrical resistance type meter). Weather conditions were measured and recorded at a permanent automatic weather station in the open and within 2 km of the fire and supplementary readings were taken in the forest during the course of the fire.

Two billy calorimeters were setup within the plot to be burnt. The billies were placed in a representative fuel-bed about 10 m inside the plot boundary. The billies were placed on the soil surface and a cable from the temperature sensing device was buried to about 2 cm below ground level in a slot made with a spade, perpendicular to the anticipated direction of the fire. The cable was buried until it reached outside the plot area to a safe site for an observer. Changes in the temperature of the billies were recorded manually. Care was taken to cause minimal disturbance to the fuel-bed when the billies were set up.

TABLE 1
Fire behaviour and weather conditions during the course of this experiment (n=35).

FIRE BEHAVIOUR VARIABLES	MEAN	MAXIMUM	MINIMUM
Forward rate of spread ($m.h^{-1}$)	35	150	5
Fine fuel load before fire ($t.ha^{-1}$)	14.7	20.9	9.5
Fine fuel load after fire ($t.ha^{-1}$)	7.5	12.3	4.2
Intensity ($kJ.m^{-1}$)[a]	169	556	25
Heat output ($MJ.m^{-2}$)[b]	15.2	31.1	3.7
Flame height (m)	0.38	0.86	0.11
Scorch height (m)	5.6	12.5	1.6
Fuel moisture content (% oven dry weight)	12.7	17.0	9.0
Proportion of fuel burnt (%)	51	73	14
WEATHER VARIABLES			
Air temperature (°C)	18	24	11
Relative humidity (%)	52	75	28
Wind speed at 10 m in open ($km.h^{-1}$)	11.9	25.8	0.6
Drought factor[c]	7.3	10.0	3.8
Soil dryness index[d] (mm equiv.)	48	136	8
Keetch Byram drought index[e] (mm equiv.)	17	102	3

[a] Fire-line intensity (Byram 1959)
[b] Heat output = available fuel energy (Byram 1959)
[c] Proportion of fine fuel available (McArthur 1973)
[d] Mount (1972)
[e] Keetch and Byram (1968)

Twelve heat sensitive plates were inserted between the fuel bed and the soil surface at uniformly spaced locations across the plot on the day of the fire. These plates were collected for analysis after the burnt area had cooled.

The upper limit of the visible scorch was measured on every scorched tree within a plot a few days after the fire. A fibreglass tape and Suunto clinometer were used to measure scorch heights. The average scorch height on a plot was used in this analysis.

The diameter of all regrowth trees and their bark thickness were measured near ground level on the burnt plots at least three to nine months after the fire. The trees were recorded as alive or fire-killed. Diameters were measured with a diameter tape and bark thickness was measured at four places around the circumference using a Gill-type bark probe (Gill *et al.* 1982). This was only done after the first rotation fires because there were insufficient trees killed by the second and third rotation fires.

Analysis

Tree diameter was plotted against bark thickness of all trees measured noting those that had died. An example of such a graph is shown in Figure 1. The threshold level of diameter and bark thickness was read off these graphs and used for the correlation and regression analysis with the billy parameters, fire behaviour variables and weather variables.

The temperature time curve of each billy calorimeter was plotted. Five parameters were determined from these curves, two of them (T_{max}, T_t) were related to the increase in temperature of the billy above the initial temperature, one (t) was the time duration of a rise in temperature at a rate greater than 1.2°C.min^{-1} and the other two (r_1, r_2) were related to the rate of temperature increase as defined in equations (1) and (2). Most temperature time curves followed the form of equation (1) but some followed equation (2). The parameters for equation (1) were determined for each billy measurement using a line of best fit approach and the parameters for equation (2) were determined where this was the best model. These six parameters were tested for linear correlation (Pearson's) with the diameter and bark thickness of the trees killed by the fires, and for correlation with the other fire and weather variables measured. A stepwise multi-regression was used to test the correlation of the diameter and bark thickness of the killed trees with the billy parameters, fire behaviour variables and weather variables.

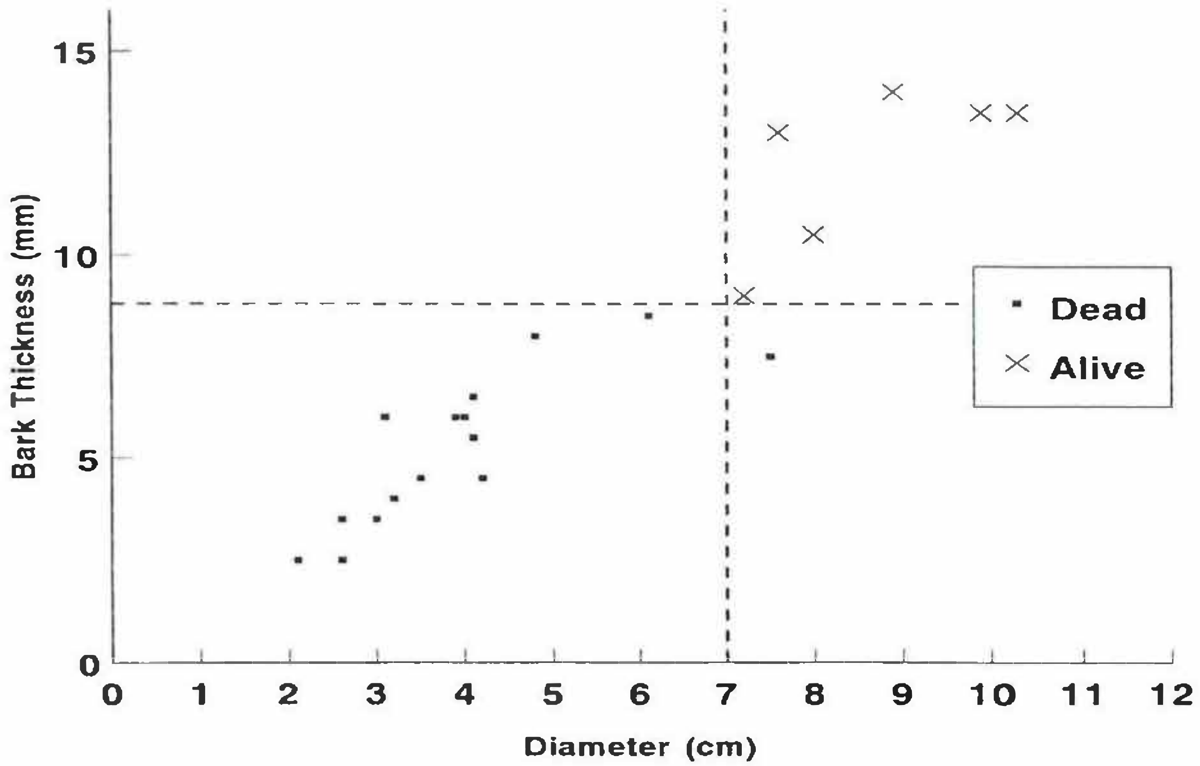

Figure 1. An example of bark thickness plotted against diameter of all trees measured on a burnt plot (35 m x 35 m) showing the delineation between the dead and alive trees and the threshold bark thickness and diameters for that particular fire. The majority of these trees were Narrow-leaf Peppermint (E. radiata).

$$T_t = \frac{T_{max}}{1 + \left(\frac{T_{max} - 0.5}{0.5}\right) e^{-r_1 t}} \quad \text{......... (1)}$$

$$T_t = T_{max}(1 - r_2^{-t}) \quad \text{.......... (2)}$$

where: T_t = temperature a time t (°C)
T_{max} = maximum increase in temperature above ambient (°C)
t = time (minutes)
r_1, r_2 = rate of increase in temperature ($°C.min^{-1}$)

The maximum temperature reached by each heat sensitive plate was determined and recorded. A graph of the proportion of plates reaching various temperatures was drawn. The mean, maximum and minimum temperature reached by a plate within each plot was recorded. The mean temperature of the plates was tested for correlation with the effects of the fire on the legume seedlings, invertebrates and soils.

RESULTS

Billy Calorimeter

The temperature time curves of the billy calorimeter followed two distinctive patterns. The first was described by the logistic equation (equation (1)), where the fire affecting the billy was predominantly a headfire with forward leaning flames (flame angle < 90°) and the second pattern was described by an asymptotic inverse log relationship described by equation (2) where the fire affecting the billy was predominantly a backfire with flames leaning away for the billy (flame angle > 90°). Examples of each of these are shown in Figures 2a and 2b respectively. In the logistic model, the temperature of the billy started to rise 2-3 minutes before the flames came in contact will the billy and rose steeply soon after the flames surrounded it. In the asymptotic inverse log model the temperature increase did not start until nearly half a minute after the flaming front had come in contact with the billy after which the rate of temperature increase was rapid and then decreased gradually.

The duration of temperature increase at a rate at least $1.2°C.min^{-1}$, averaged 6.1 minutes (se=0.44, n=59), but lasted for up to 14 minutes, which was long after the flaming front had passed. This duration of heating tended to be shorter for headfires ($\overline{x}$=5.7 min., se=0.53, n=43) and longer for the backfires ($\overline{x}$=7.0 min., se=0.65, n=16), but this difference was not statistically significant.

The linear correlations between the diameter and bark thickness of the trees killed by each fire are given in Table 2. The only correlation that is significant is that between tree diameter and the increase in temperature above ambient (T_{max} in equations (1) and (2)). T_{max} was most strongly correlated with the total heat output from the fire (Pearson's correlation coefficient = 0.44, Pr = 0.0004) which is directly related to the amount of fuel burnt. The heat output of the fire was more strongly correlated to the diameter of the trees killed than the temperature increase in the billy, T_{max}, (Pearson's correlation coefficient of 0.73, Pr = 0.0001).

(a) Headfire (b) Backing fire

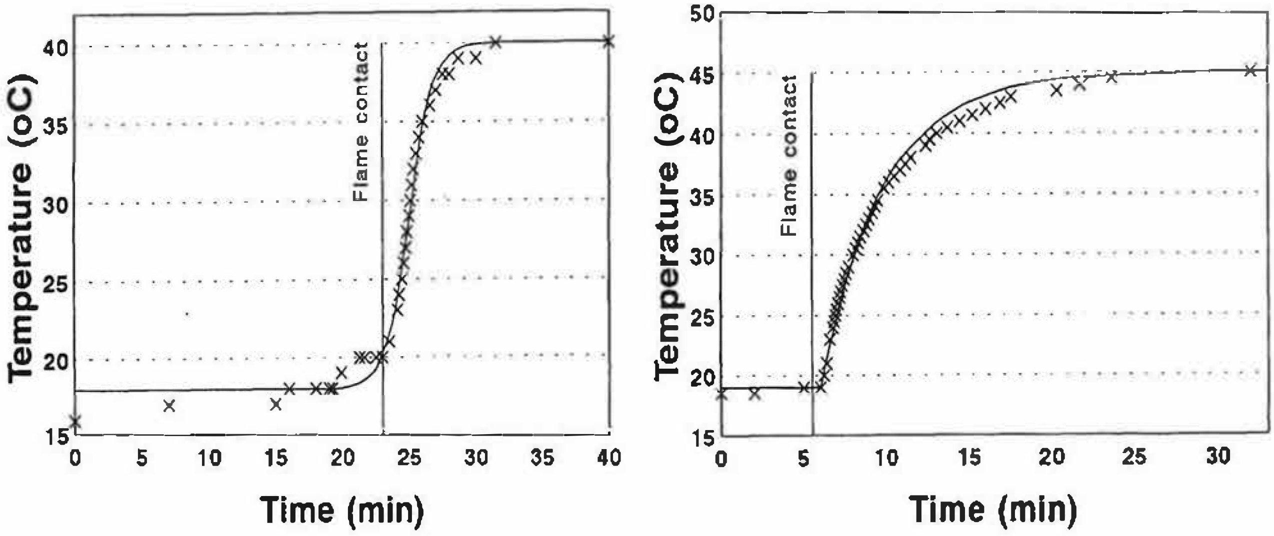

Figure 2. An example of the temperature time response curve for a billy calorimeter (a) in front of a headfire, and (b) in front of a back fire. The observed points are marked individually (x) and the predicted models are shown as a solid line (—) (a) is equation (1) and (b) is equation (2).

TABLE 2

Pearson's linear correlation coefficient between the threshold tree diameter and bark thickness of trees killed by fire. The probability of the correlation being greater than zero is given in brackets and n=33 for all correlations.

	TEMPERATURE INCREASE ABOVE AMBIENT (T_{max})	DURATION OF HEATING AT 1.2°C.min^{-1} OR FASTER	RATE OF HEATING (r_1 IN EQUATION (1))
Diameter	0.36	0.28	0.04
	(0.038)	(0.113)	(0.819)
Bark Thickness	0.07	0.05	0.03
	(0.711)	(0.793)	(0.856)

When all fire behaviour, weather and billy calorimeter variables were included in a stepwise multiple linear regression analysis, none of the billy variables were included in the final regression. The threshold diameter and bark thickness of killed trees was best predicted by the relationships:

Threshold Diameter = 0.21 Heat Output - 0.52 Fuel Moisture - 0.29 Scorch Ht + 0.37 KBDI + 9.25

(r^2=0.80, n=33, Pr = 0.0001)

Data range: Threshold Diameter, 4.5 to 10.5 cm
Heat Output, 3.7 to 31.1 MJ m^{-2}
Fuel Moisture, 9.0 to 17.0 per cent oven dry weight
Scorch Height, 1.6 to 12.5 m
KBDI, 3.1 to 101.8 (mm equivalents)

Threshold Bark Thickness = 3.72 Flame Height + 0.048 Heat Output + 5.67

(r^2=0.46, n=33, Pr = 0.0001)

Data range: Threshold Bark Thickness, 4.0 to 9.7 mm
Flame Height, 0.1 to 0.9 m
Heat Output, 3.7 to 31.1 MJ m^{-2}

The above regressions and Table 2 show that it was easier to predict the threshold diameter of the trees killed by the fire, under the experimental conditions experienced, than the threshold bark thickness. This may be due to less variability in the diameter measurements and reflect better the bark thickness of the trees before the fire and it may be due in part to a greater ability of larger trees to dissipate heat.

Using the measured fire behaviour and environmental variables, the maximum temperature rise and the rate of this rise in the billy calorimeter were best described by the following regressions:

T_{max} = 1.06 Heat Output - 0.19 KBDI - 16.03 Flame Height + 17.94

(r^2=0.30, n=33, Pr = 0.0006)

Data range: T_{max}, 6 to 60°C
Heat Output, 3.7 to 31.1 MJ m^{-2}
KBDI, 3.1 to 101.8 (mm equivalents)
Flame Height, 0.1 to 0.9 m

r_1 = -0.082 Fuel Moisture + 1.66

(r^2=0.19, n=33, Pr = 0.0005)

Data range: r_1, 0.15 to 2.00°C.min^{-1}
Fuel Moisture, 9.0 to 17.0 per cent oven dry weight

The temperature rise in the billy can be used to calculate the amount of heat absorbed. The billies described here have a water capacity of 2800 g, a basal area of 177 cm^2 and a height of 17 cm. Since the heat capacity of water is 4.1868 J.deg^{-1}.g^{-1}, then a 1°C rise in temperature of the billy is equivalent to 662 kJ.m^{-2}. If the water in the billy was to rise by 40°C, the the heat absorbed would be equivalent to 26.5 MJ.m^{-2}, about 177 per cent of the expected heat yield from a low intensity fire of 15 MJ.m^{-2}. This indicates that the heat catchment of the billy can be at least twice the area occupied by the billy.

Heat Sensitive Plates

The heat sensitive plates gave a spatial measure of the temperature variability between the litter and the mineral soil surface. The average, maximum and minimum temperatures for each fire are summarized in Table 3. Surface soil temperatures were much lower in the first rotation (1R) spring fires compared with the second (2R) and third rotation (3R) fires and the autumn fires. The 2R and 3R spring fires applied just as much or more heat to the soil surface as the autumn fires, but the 2R autumn fires heated the surface soil less than the 1R autumn fires.

TABLE 3

Summary statistics of surface soil temperature for two different seasons and three different rotations (1R, 2R, 3R). Temperatures are in °C and the standard errors of the means are shown in parentheses. All values are based on the average of 12 observations per fire. Fire rotations were 2-4 years apart.

	SPRING 1R	SPRING 2R	SPRING 3R	AUTUMN 1R	AUTUMN 2R
Mean	161	403	427	367	292
	(30)	*(14)*	*(55)*	*(39)*	*(40)*
Maximum	340	453	577	530	396
Minimum	83	370	239	185	149

The spatial variability of the surface soil heating can be better seen in Figures 3 a, b, c, d and e using the method of presentation suggested by Hobbs *et al.* (1984). Figure 3a shows that the variability in surface soil temperature in the 1R spring fires was relatively low, with few areas being heated above 300°C, but most the area was heated to about 100°C. This was in contrast to the 2R spring fires shown in Figure 3b where most of the soils were heated to 350°C, but few higher than 500°C. The 3R spring fires (Fig. 3c) were quite variable and there was a fairly even distribution of temperatures across the burnt area, indicating a patchier burn. Figure 3d shows that the 1R autumn fires, like the 1R spring fires, were not especially variable either, however, the temperatures reached at the soil surface were generally above 400°C. As with the 3R spring fires, the 2R autumn fires (Fig. 3e) were patchy and showed a wide range in temperatures.

The surface soil temperature can be combined with the moisture content of the soil at the time of burning with the diffusivity of the soil (diffusivity = conductivity/capacity) to provide an indication of the depth and amount of soil heating. A model such as that proposed by Aston and Gill (1976) or Campbell *et al.* (1990) can be used to predict soil temperature profiles in time. The moisture content of the soil can be estimated using a modified form of Mount's soil dryness index (Mount 1972). The modification of this index is to invert it to become a 'soil moisture index' and to express this as a proportion of 50 mm rather than the usual 200 mm. This last modification more closely reflects the average moisture content of the top 10 cm of soil. Figure 4 shows how this soil moisture index relates to gravimetrically determined soil moistures.

Combining data from the vegetation project and the fire behaviour project described in Tolhurst and Flinn (1992), a clear association is apparent between the increase in legume density following fire and the surface soil temperature. A summary of the results is shown in Table 4. This shows that even though the fire intensity and the total heat output from the fires in spring and autumn were similar, the average surface soil temperatures were very different and the increase in legume density was most pronounced with the higher indicated surface soil temperatures. This observation concurs with the conclusion of Christensen and Kimber (1975) that the percentage of fuel consumed and hence the degree of exposure of mineral soil is the more important than the fire intensity *per se* or the amount of heat output from the fires.

The heat sensitive plates also give a measure of heat transfer to the soil. The heat capacity of aluminium is 0.90 $J.deg^{-1}.g^{-1}$, each plate weighs approximately 12 g and covers an area of about 60 cm^2, therefore the amount of heat absorbed by a single plate is 1.8 $kJ.deg^{-1}.m^{-2}$ and the heat input into the soil is therefore a product of 1.8 times the maximum temperature of the plate. If the total heat output from the fire is about 15 $MJ.m^{-2}$, then the proportion of heat being transferred from the fire to the soil is directly related to the maximum temperature of the plate. Table 5 indicates the amount and proportion of heat transferred to the soil as measured by a heat sensitive plate described here.

TABLE 4

Average fire intensity, total heat output, surface soil temperature and increase in legume density associated with spring and autumn fires in the Blakeville Fire Effects Study Area, Wombat State Forest, Victoria.

SEASON	FIRE INTENSITY ($kW.m^{-1}$)	HEAT OUTPUT $MJ.m^{-2}$)	SURFACE SOIL TEMPERATURE (°C)	FACTOR OF LEGUME DENSITY INCREASE (POST-FIRE/PRE-FIRE)
Spring	205	16.9	78	6.0
Autumn	156	15.4	315	12.3

TABLE 5

Relationship between maximum temperature of a heat sensitive plate and the amount of heat absorbed and the proportion of the total heat released by a 15 MJ.m² fire.

Maximum Temperature (°C)	100	200	300	400	500	600
Heat Input to Soil (kJ.m⁻²)	180	360	540	720	900	1080
Percentage of Total Fire Heat	1.2	2.4	3.6	4.8	6.0	7.2

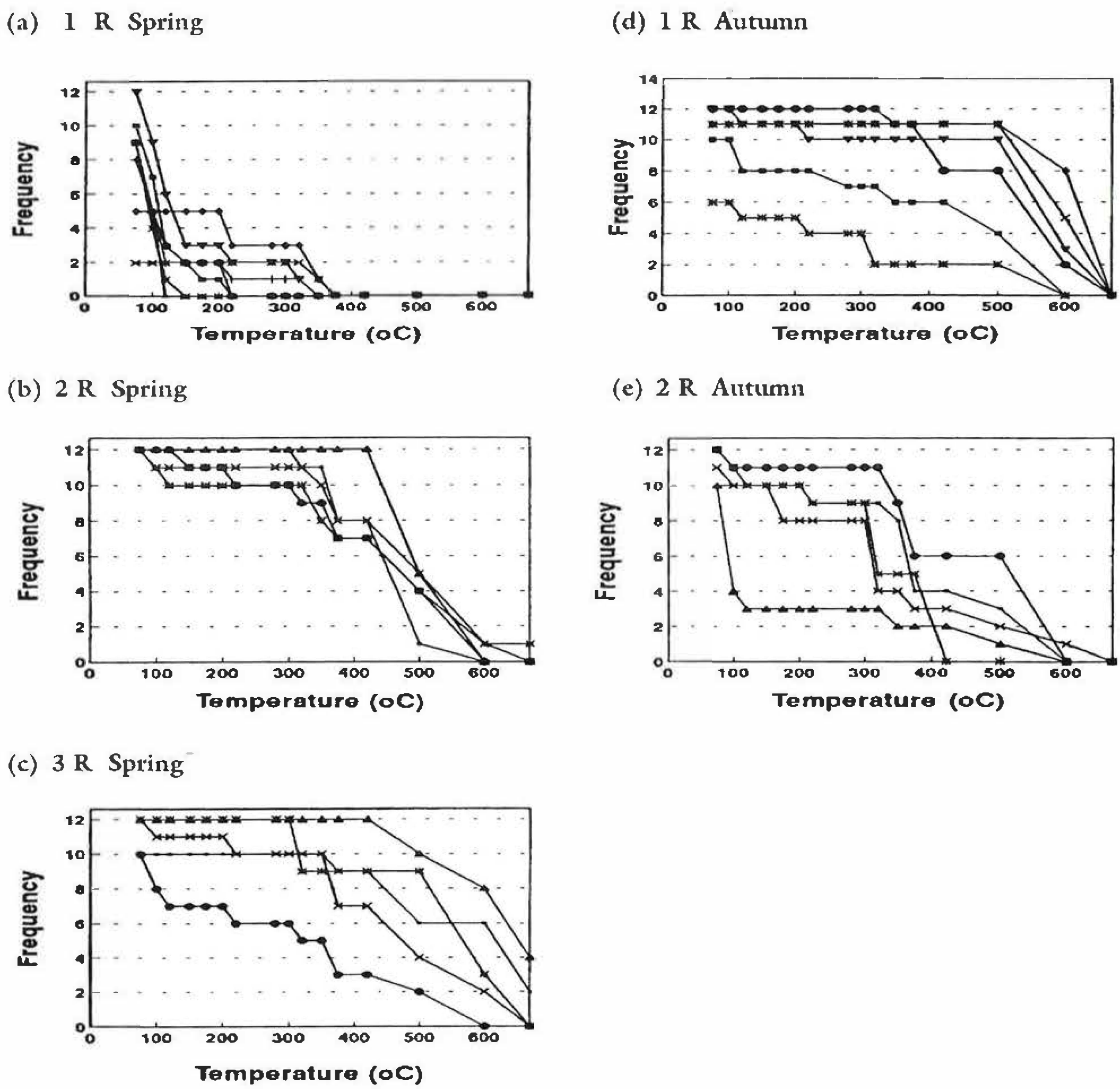

Figure 3. Frequency distribution of temperatures reached at the soil's surface during spring and autumn burning with up to three successive fires. Temperatures were measured with heat sensitive plates. Each line represents one fire replicate. (a) is the 1R spring fires, (b) is the 2R spring fires, (c) is the 3R spring fires, (d) is the 1R autumn fires and (e) is the 2R autumn fires.

Scorch Height

The third measure of sensitive heat is indicated by the trees on the site. Scorching of the leaves results from the interaction of the temperature of the air in the vicinity of the leaves, the amount of heat being released from the fire and the cooling rate of transpiration.

The scorch height of the fires in spring was found to be about 4 to 6 m, the scorch height in autumn was 7 to 12 m even though the average fire intensities in spring and autumn were similar (150 kJ.m^{-1}, Tolhurst and Flinn 1992). Average scorch height was best predicted by the amount of fine fuel burnt in combination with the air temperature within the forest and the Keetch Byram drought index (KBDI). The relationship between these variables was described by the function:

DISCUSSION

Billy Calorimeter - a Flaming Zone Pyranometer

The billy calorimeter used here proved successful in providing information which differentiated between fires. The total amount of heat absorbed by the billy reflected the amount of heat released from the fuel, as was found by Beaufait (1966) and Knight (1981), but it was also related to the Keetch Byram Drought Index and average flame height. However, because the fuel had been sampled from 15 points within the plot and the billy only sampled one point, it could be expected that the amount of fuel burnt would give a better average heat output for the plot and hence be better correlated with the threshold diameters and bark

$$\text{Scorch Ht.} = 0.631\ \text{Fine Fuel Burnt} + 0.375\ \text{Air Temperature} + 0.052\ \text{KBDI} - 6.977$$

$$(r^2=0.64,\ n=25,\ Pr = 0.0001)$$

where: Scorch Ht = average scorch height (m) (data range, 1.6 to 12.5 m)
Fine Fuel Burnt = (t ha^{-1}) (data range, 2.0 to 14.7 t ha^{-1})
Air Temperature = (°C) (data range, 12.0 to 24.0°C)
KBDI = Keetch Byram drought index (mm equivalents) (data range, 3.0 to 57.0)

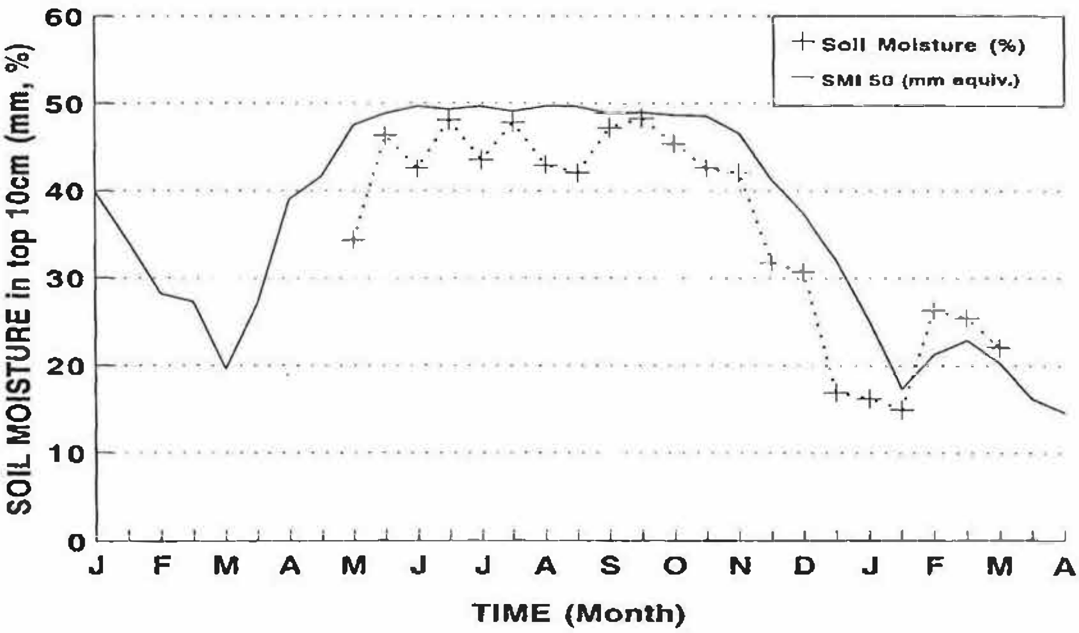

Figure 4. Seasonal variation in gravimetrically determined soil moisture content of the top 10 cm of soil and the soil moisture index calculated using meteorological data.

thickness of fire killed trees. More billies may improve their correlation. The rate at which the heat was absorbed by the billies during the fire varied between fires, but this could not be related to the mortality of the trees within the plot. The rate of heat absorbed by the billy was significantly, but weakly negatively correlated with fine fuel moisture content, which is probably due to a greater rate of combustion when the fuels are drier (Cheney 1981).

The response curve obtained from the billy was similar in shape to that reported by Vines (1968) for the temperature of a jarrah (*Eucalyptus marginata* Sm.) tree bole at the cambium layer when exposed to a 'fierce' fire, it was also similar to those reported by Gill and Ashton (1968) when exposing bark to a radiant heater in the laboratory, and those obtained by Fahnestock and Hare (1964) in forest fires, indicating that the billy calorimeter is a reasonable analogue of a tree. The main difference between the billy and a tree is that there is no cooling mechanism in the billy and therefore the temperature falls much slower than it does in a tree. Trees are able to conduct heat away from the base of the tree through sap movement associated with transpiration at longitudinal movement rates of up to 15 $cm.min^{-1}$ (Doley and Grieve 1966). Another difference between the billy and a tree bole is the height of the billy. Hare (1965) observed that under windy conditions, a convection column on the lee of the tree increases the heat load on the bole. There was little opportunity for this phenomenon to develop with a billy. However, for the limited set of fire and weather conditions experienced in this experiment, the total heat output from the fire, the fine fuel moisture content, average scorch height and the Keetch Byram Drought Index were most closely correlated with the size of the trees killed by the fire.

The experimental fires in which the billy calorimeter has been used to date have been in natural fuel beds. In this situation the amount of sensible heat has been strongly correlated to the amount of fuel burnt. Generally, the more fuel burnt the longer the period of heating and so no additional information is provided by knowing the duration or rate of heating. These conditions may change if wind or slope are more dominant in the fire behaviour, as a greater proportion of the heat may move along close to the ground surface before rising and therefore generate a greater amount of sensible heat in the flaming zone. Because the moisture content of the fuel may affect the rate of combustion, it may be expected to affect the amount of heat released, but usually the conditions under which fuel moisture are higher will coincide with conditions when a smaller percentage of the total fuel will be burnt.

Although there was a strong linear relationship between the bark thickness of the trees and diameter, the threshold diameter for tree mortality was more easily predicted than the threshold bark thickness. In functional terms bark thickness is a better variable to measure because Gill and Ashton (1968), Vines (1968), and Hare (1965) have shown that bark thickness is the primary factor in determining whether or not the cambium is exposed to lethal temperatures for a given fire intensity or heat exposure, but this could be used to advantage. In this study, we found that the thickness of bark killed on large trees was similar to the threshold bark thickness of fire killed trees. Gill *et al.* (1986) also found that the amount of bark killed on gums was related to the period of flaming. Where gum barked trees occur, the thickness of the bark killed by the fire could be used as a direct measure of sensible heat output from the fire, solar radiation, wind effects and ambient air temperature, but this could not be measured until a number of months after a fire when decortication occurs.

Heat Sensitive Materials - Below Flaming Zone Pyranometers

The heat sensitive plates show a variation in the fires not apparent in the fire intensity or heat output of the fires. The patchiness of the 2R autumn fires and the 3R spring fires are evident from the frequency graphs and this can be translated into patchiness in the effects the fire will have on the soils, plants and invertebrates. The contrast in the 1R spring and 1R autumn fires is not apparent when viewing fire intensity data, but the surface soil temperature differences are marked. This has had a significant affect on the legume seedling regeneration which will later be reflected in a difference in the structure of the understorey. The heat sensitive plates have therefore been useful in identifying differences in fires at an ecological level.

The temperature profile of the soil is related to the surface temperature of the soil, its heat capacity, and its thermal conductivity properties, these in turn are affected by the density and moisture content of the soil. Scotter (1970) and Raison *et al.* (1986) found a strong relationship between the surface soil temperature and the temperature profile induced by forest fires. The surface soil temperature as measured by the heat sensitive plates could therefore be used in conjunction with estimates of soil moisture and thermal conductivity to estimate the temperature profile induced by a fire.

Data from the heat sensitive plates can be used directly to correlate with the ecological effects of fire. Noble (1984) found a strong linear relationship between the proportion of eucalypt lignotubers surviving a fire and the surface soil temperature measured with this technique. Collett *et al.* (1993) found a strong correlation between the surface soil temperature measured during a fire and the subsequent abundance of annelids (earth-worms) for successive fires on the same site even though the intensity of the two fires were similar. Bentley and Fenner (1958) used heat sensitive plates to measure the soil temperature profiles in grass and shrub fuel and found it an objective way to classify seedbeds. I would also expect that the surface soil temperatures could be used to predict the patterns of post fire germination of legumes as reported by Auld

(1986) and Auld and O'Connell (1991) in the same way as the data presented here shows.

The amount of heat absorbed by the plate also indicates the proportion of heat being directed to the soil. Because the heat capacity and weight of the material being used for the plate is known, the indicated temperature can be used to calculate the amount of heat absorbed so, for example, if a fire yields 15 000 kJ.m^{-2} of energy, and the plate reaches 400°C, then the proportion of heat absorbed is about 4.8 per cent which agrees with previous published results. Raison *et al.* (1986) found that between 4.5 per cent and 6.9 per cent of the heat released from the surface fuel in a low intensity forest fire was transferred to the soil, Packham (1970) calculated that about 5 per cent of the energy released is transferred to the soil and DeBano *et al.* (1977) suggested that only about 8 per cent of the heat was transferred into the soil. The amount of heat transferred to the soil will be a function of the amount of heat released by the fire, and the proportion of the surface fuel burnt and hence the amount left behind to insulate the soil. Heat sensitive plates integrate all these factors well. The advantage of heat sensitive plates is that it gives a good indication of the variability of the fire which is difficult to get with measurements of fire intensity.

Hobbs and Gimingham (1984) used a combination of thermocouples and heat sensitive plates in their study of heath fires. They found that the maximum temperature reached in any fire showed a similar pattern in relation to the fuel structure as did the duration of heating over 400°C. This indicated that very little information was lost by using the maximum temperature alone. Fire intensity was also measured in these fires and it was found that the relationship between fuel structure and intensity was different to that of temperature and fuel structure. The authors expressed the view that 'direct temperature measurements may provide more meaningful information' than fire intensity values.

Scorch Height - Above Flaming Zone Pyranometers

The measurement of scorch height is an end in itself. The canopy of the overstorey is important to arboreal birds, mammals and invertebrates who rely on the canopy for food. Reduction in the canopy can also lead to reduced timber production or perhaps timber degrade (Kellas *et al.* 1984).

Scorch height has been found to be related to the ambient air temperature, wind speed and flame height (Van Wagner 1973; Cheney *et al.* 1992; Buckley 1993). Scorching of the canopy occurs when leaf tissues can no longer cope with the heat load of the fire. Therefore, scorching is dependent on the characteristics of the fire, the air environment of the forest and the structural and physiological status of the trees.

In this study, scorch height was found to be significantly related to the heat output of the fire, the ambient air temperature and the seasonal dryness as indicated by the Keetch Byram Drought index. Because the tree canopy is above the surface fire, it has to cope with the convective heat produced by the burning litter. In higher intensity fires and fires with a shrubby understorey, radiation may become more important. The ambient air temperature is important because it determines what increase in temperature is needed for lethal temperatures to be reached. Ambient air temperature may also affect the rate of combustion and affect the relative humidity of the air. The KBDI affects the ability of the canopy to cope with the heat load. When the KBDI is high, there will be greater moisture stress on the trees and therefore they will be less able to dissipate heat as rapidly through transpiration. Scorch height is therefore a useful measure of sensible heat output above the flaming zone, integrating fire and seasonal conditions.

A useful result from this work is the finding that KBDI is an important predicting variable. The observations by Wallace (1966) that scorch height in jarrah (***Eucalyptus marginata*** Sm.) forest was seven times flame height in spring and 14 times flame height in autumn can be reflected in the Keetch Byram Drought Index. Scorch height can therefore be predicted with more precision than that provided by the Forest Fire Danger Meter (McArthur 1973).

CONCLUSIONS

The billy calorimeter was used to measure the amount and rate of sensible heat transfer in the flaming zone of a surface forest fires. Two distinct temperature profiles were described for the rate of temperature increase in the billy calorimeter. The first followed a logistic function which described the temperature profile when the billy was exposed to a headfire, and the second was an asymptotic inverse logarithmic function which applied when the billy was exposed to a backfire. The maximum increase in the billy temperature (T_{max}), was related to the total heat output of the fire, the seasonal dryness as measured by the Keetch Byram drought index and flame height. The rate of temperature increase was weakly related to the ambient air temperature.

The most useful information derived from the billy calorimeter was the total temperature rise above ambient. While this was statistically related to the threshold diameter of the trees killed by the fires measured, the amount of fuel burnt was the primary factor affecting tree death. The rate of temperature increase of the billy was not significantly related to tree death in this study. Further investigations are needed to determine if these same conclusions are true under high intensity fires and when wind or slope are more dominant factors. Where gum barked trees exist, the thickness of the bark killed by a fire is a useful measure of sensible heat output, but this cannot be measured until several months after the fire.

Heat sensitive plates were used to measure the amount of sensible heat transferred to the soil below the flaming zone of a surface forest fire. Maximum surface soil temperatures could be used in conjunction with estimates of soil moisture content and soil conductivity to estimate soil temperature profiles resulting from fires. The maximum surface soil temperatures in themselves were correlated with the survival of earthworms, establishment of legume seedlings, and death of lignotubers. Heat sensitive plates were useful for indicating the amount of variability in a fire which is not evident from measures of fire behaviour such as fire intensity.

Tree scorch height was used as a measure of the amount of sensible heat transferred above the flaming zone of a surface forest fire. This measure was reliant on there being an uneven-aged forest so that scorch height could be measured when it was less than canopy levels. Scorch height was easy to determine and was directly related to the ecological impacts of fire on tree growth, and arboreal animal food supply.

Measures of sensible heat integrate fire, weather, topographic and seasonal conditions as well as heat transfer processes. They therefore offer an efficient way of quantifying fires in an ecologically meaningful way.

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APPENDIX 1

TIME IN MINUTES

°C	CRAYON	0	1	2	3	4	5	7	10	15	20
	670	GREEN	DARKER	DARKER	GREY/GREEN	LT/GREY					
	600	BLUE	AQUA	WHT/AQUA	WHITE	WHITE			MELT DOWN OF FOIL →		
670	500	LT/BROWN	DK/BROWN	PINK	SAME	***					
	420	WHITE	YELLOW	GREY							
	350	BROWN	RED/BROWN	SAME							
	670	GREEN	N.C.	N.C.	DARKER	SAME	SAME	WHT/GREY	GREY	SAME	SAME
	600	BLUE	N.C.	N.C.	LIGHTER	LIGHTER	WHT/AQUA	WHITE	SAME	SAME	SAME
600	500	LT/BROWN	DARKER	DARKER	DK/BROWN	SAME	RED/GREY	SAME	SAME	PINK	
	420	WHITE	YELLOW	YELL/BROWN	YEL/GREY	SAME	SAME	SAME	YELLOW	SAME	WHITE
	350	BROWN	RED/BROWN	DARKER	DARKER	SAME	BLACK				
	600	BLUE	N.C.	N.C.	N.C.	LIGHTER	SAME	LIGHTER	SAME	WHITE	SAME
	500	LT/BROWN	N.C.	DARKER	DARKER	N.C.	DARKER	DK/BROWN	DK/BROWN	SAME	SAME
500	420	WHITE	WHT/YELLOW	YELLOW	YEL/BROWN	YEL/BROWN	SAME	SAME	SAME		
	350	BROWN	DARKER	RED/BROWN	SAME	SAME	***				
	320	GREEN	LT/BLUE	SAME	FAWN	SAME	***				
	500	LT/BROWN	N.C.	DARKER	DARKER	DARKER	SAME	SAME	DK/BROWN	DK/BROWN	SAME
	420	WHITE	N.C.	SAME	CREAM	SAME	WHT/YELLOW	YELLOW	DARKER	YEL/BROWN	YEL/BROWN
420	350	BROWN	DARKER	DK/BROWN	RED/BROWN	SAME	SAME	DK/BROWN	SAME	SAME	SAME
	320	GREEN	LIGHTER	LT/BLUE	LIGHTER	SAME	SAME	GREY/BLUE	SAME	FAWN	FN/GREEN
	300	PL/GREEN	OLV/GREEN	DARKER	***	SAME	SAME	SAME	SAME	SAME	
	420	WHITE	N.C.	N.C.	DARKER	SAME	SAME	SAME	SAME	WHT/YELLOW	YELL/BROWN
	350	BROWN	N.C.	LIGHTER	SAME	LIGHTER	SAME	SAME	SAME	RED/BROWN	SAME
350	320	GREEN	N.C.	DK/GREEN	LT/BLUE	GREY/BLUE	GREY	SAME	SAME	SAME	DK/GREEN
	300	PL/GREEN	DARKER	BRN/GREEN	DARKER	SAME	SAME				
	280	LT/GREEN	OLV/GREEN	BLACK	BLACK	SAME	SAME				
	320	GREEN	N.C.	N.C.	N.C	N.C.	LIGHTER	LT/BLUE	SAME	GREY/BLUE	SAME
	300	PL/GREEN	DARKER	DARKER	SAME	DARKER	DARKER	BRN/GREEN	SAME	OLV/GREEN	SAME
300	260	LT/GREEN	DARKER	SAME	SAME	SAME	GREEN/GREY	GREY	SAME	SAME	***
	220	WHITE	DARKER	SAME	SAME	DARKER	YEL/BROWN	YEL/BROWN	LT/BROWN	DARKER	***
	200	DK/BLUE	DK/GREEN	BLACK	SAME	***					
	220	WHITE	N.C.	N.C	N.C.	N.C.	N.C.	DARKER	DARKER	LT/YELLOW	LT/YELLOW
	200	DK/BLUE	DARKER	DARKER	DARKER	DARKER	DARKER	DARKER	DARKER	BLACK	
200	150	FN/GREEN	DARKER	FAWN	PURPLE	DARKER	SAME	***			
	120	CREAM	FAWN	DARKER	FN/GREEN	SAME	SAME	***			
	100	FAWN	PINK	BLUE	SAME	SAME	***				
	150	FN/GREEN	N.C.	N.C.	N.C.	N.C.	GREENER	GREENER	FAWN	FAWN	PURPLE
	120	CREAM	N.C.	N.C.	N.C.	DARKER	FN/GREEN	VIOLET	SAME	***	
150	100	FAWN	N.C.	DARKER	BLUE	SAME	***				
	75	CRM/GREEN	GREENER	DARKER	DARKER	AQUA	SAME	***			
	65	FN/GREEN	FAWN	OLV/GREEN	BLUE/GREEN	SAME	***				
	150	FN/GREEN	N.C.	***							
	120	CREAM	N.C.	N.C.	N.C.	N.C.	N.C.	N.C.	N.C.	FN/GREEN	VIOLET
100	100	FAWN	N.C.	N.C.	N.C.	N.C.	N.C.	N.C.	DARKER	BLUE	SAME
	75	CRM/GREEN	N.C.	DARKER	DARKER	AQUA	SAME	***			
	65	FN/GREEN	N.C.	OLV/GREEN	OLV/GREEN	BLUE/GREEN	SAME	***			
	150	FN/GREEN	N.C.	***							
	120	CREAM	N.C.	***							
75	100	FAWN	N.C.	***							
	75	CRM/GREEN	N.C.	N.C.	DARKER	DARKER	AQUA	SAME	***		
	65	FN/GREEN	N.C.	N.C.	OLV/GREEN	BLUE/GREEN	SAME	***			

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A framework for assessing acute impacts of fire in jarrah forests for ecological studies

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ABSTRACT

Bushfires are commonly described in terms such as rate of spread, flame dimensions or Byram's intensity, which reflect suppression difficulty or general damage potential. Meaningful descriptors of the fire environment for interpreting ecological effects are those which reflect the amount and rate of heat energy released and its distribution. These factors determine the immediate physical or acute impacts of fire which give rise to ecological effects.

The acute impacts of fire in jarrah forests can be studied and predictive models developed by:

(i) stratifying the area within and around the flames (impact zones);
(ii) identifying the physical impacts within these zones;
(iii) identifying readily measurable descriptors of the amount and rate of heat energy release;
(iv) identifying factors affecting the transfer of heat to plant tissue and the soil;
(v) experimenting to develop functional relationships between physical impacts, readily measurable fire descriptors and factors affecting heat transfer.

INTRODUCTION

Forest fires are described according to the level of interest and knowledge of the observer. Fire managers and fire behaviour scientists commonly describe fire in terms that convey information about the difficulty of suppression or the damage potential of the fire, such as linear and area rates of spread, flame dimensions, flame residence time and intensities, and there is ample literature which defines, models, and describes ways of measuring or calculating these variables (e.g., Davis 1959; Luke and McArthur 1978; Gill *et al.* 1981; Chandler *et al.* 1983; Gill and Knight 1988; Johnson 1992).

When studying fire effects on biotic and abiotic ecosystem components, it is important to identify and measure variables which are linked to the immediate impacts of fire. Immediate or acute impacts are defined here as the physical impacts of fire on the ecosystem components imparted during the flaming and smouldering phases of combustion. For jarrah forest fuels, most acute impacts occur over 1-2 minutes during the passage of flames. Friend (1993), in reviewing the effects of fire on small vertebrates, used the definition of acute impact provided by Warren *et al.* (1987), which includes the combustion phase and the time to the commencement of vegetative regeneration, which could be several months after the fire.

Acute impacts give rise to ecological responses, but fire ecologists have generally displayed indifference to how fires actually produce their ecological effects (Johnson 1992). McArthur and Cheney (1966) noted that in a literature review of fire effects by Hare (1961) there was an absence of a precise description of the type of fire causing the effects. Alexander (1982) reported that this trend had continued in spite of advances in the science of fire behaviour.

As well as describing fires for interpreting ecological effects, it is equally important to be able to link fire variables, factors affecting heat transfer and acute impacts so that managed fire can be effectively and reliably applied to achieve a desired ecological outcome. Managing fire in jarrah (*Eucalyptus marginata*) forests of south-west Western Australia includes prescribing fires to reduce fuel levels without damaging the boles and crowns of trees, and prescribing fires to regenerate or control specific plant species.

An understanding of fire-induced ecological responses cannot be gained by simply measuring common fire behaviour descriptors and it is unlikely that the acute impacts of fire will be modelled from first principles in the foreseeable future for several reasons. Firstly, combustion and heat transfer are separate processes, both of which are complex and poorly understood. Secondly, there is limited information linking fire behaviour variables with acute impacts, one exception being the linkage between fire intensity, flame dimensions and scorch height (Van Wagner 1973; Luke and McArthur 1978; Cheney *et al.* 1992). Thirdly, ecosystem responses will depend on numerous other interrelating factors including climate, landform and soils, the fire regime (Gill 1977, 1981), fire size, patchiness, the ecological and biological characteristics of the ecosystem (e.g., life cycles, regeneration and re-colonization strategies, adaptive traits expressed by organisms), and precedent and antecedent weather.

Therefore, in the absence of universal physical models, empirical or semi-empirical models which predict acute impacts from readily measured fire behaviour variables and heat transfer factors will continue to be developed, or existing ones validated for each vegetation type or fuel complex. This paper presents a framework for assessing the acute impacts of forest fires which give rise to ecological responses and to commercial losses.

ACUTE IMPACTS OF FIRE

The precise nature and severity of acute impact varies considerably from one fire circumstance to another, but generally acute impacts can be characterized as:

(i) reduction or removal of live and dead vegetation resulting in changes in cover, structure and habitat to varying degrees;
(ii) some plant and animal death and injury; and
(iii) soil heating and subsequent effects on soil chemistry, structure and various soil borne organisms.

Acute impacts on the vegetation equates to an almost instantaneous change in habitat (food, shelter and breeding sites), particularly if the fire is intense. Thus, the impact on fauna will depend on habitat requirements, the extent to which these have been affected by the fire, and on the biology of various taxa (Friend 1993). Therefore, measuring the impact of fire on vegetation is the key to interpreting impact on fauna.

The extent or severity of acute impact on an ecosystem will depend on the amount and rate of heat energy released and on the amount and rate of heat transfer to plants and the soil. Flames are the essence of a bushfire. Flame temperature *per se* does not necessarily relate to the amount of heat given off as temperature is a measure of the degree of hotness while heat is the quantification of the work transferred from a warm body to a cool body. However, the temperature history experienced by plant tissues does relate to thermal death time (Wright 1970; Engle *et al.* 1989) so intuitively relates to the threat posed by fire to plant tissue. For example, Ryan (1982) found that Douglas Fir (*Pseudotsuga menziesii*) seedlings can tolerate a temperature of 50°C for 1 hour, 60°C for 1 minute and 70°C for 1 second.

It is therefore important to measure and describe the fire and factors affecting heat transfer in terms that best reflect or characterize the temperature histories experienced by plant tissue and the soil at critical locations. Measuring the temperature histories of bushfires to characterize fire intensity has met with mixed success, mainly because of the high degree of temporal and spatial variation in temperature (e.g., see Hobbs and Atkins 1988; Cheney *et al.* 1992; Moore and Gill, this conference). There are also practical difficulties with using this technique beyond small, intensively monitored experimental fires. However, thermocouple temperature histories can be a useful means of correlating temperature histories with meaningful and more easily measurable fire descriptors such as flame dimensions and measures of fire intensity (see Moore, Gill and Tolhurst, this conference). Examples of thermocouple temperature histories recorded during an experimental jarrah forest fire are shown in Figure 1. These are typical of temperature histories reported by many workers for other fuel types (e.g., Pompe and Vines 1966; Rothermel 1972).

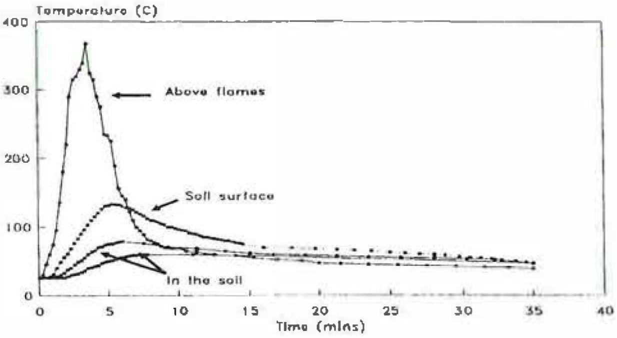


Figure 1. An example of thermocouple temperature histories at various locales during a jarrah forest fire.

ACUTE IMPACT ZONES

I present a framework which I have found useful for describing the acute impacts of fire which give rise to ecological responses in the jarrah forest. This framework, shown in a broader context in Figure 2, is comprised of four elements:

(i) Biophysical, biological and ecological information;
(ii) Fire regime information;
(iv) Factors affecting combustion and heat transfer;
(v) Physical, acute impacts of fire on the biota.

No attempt is made to predict temperature histories at a particular locale. The notion is that temperature history, therefore the acute impacts of fire, are related to fire behaviour variables which reflect heat output, and factors affecting heat transfer; variables which are more readily measurable in the field than temperature histories.

The approach I have taken for jarrah forest fires is to:

(i) stratify the area within and around the combustion zone;
(ii) identify important physical impacts within these strata;

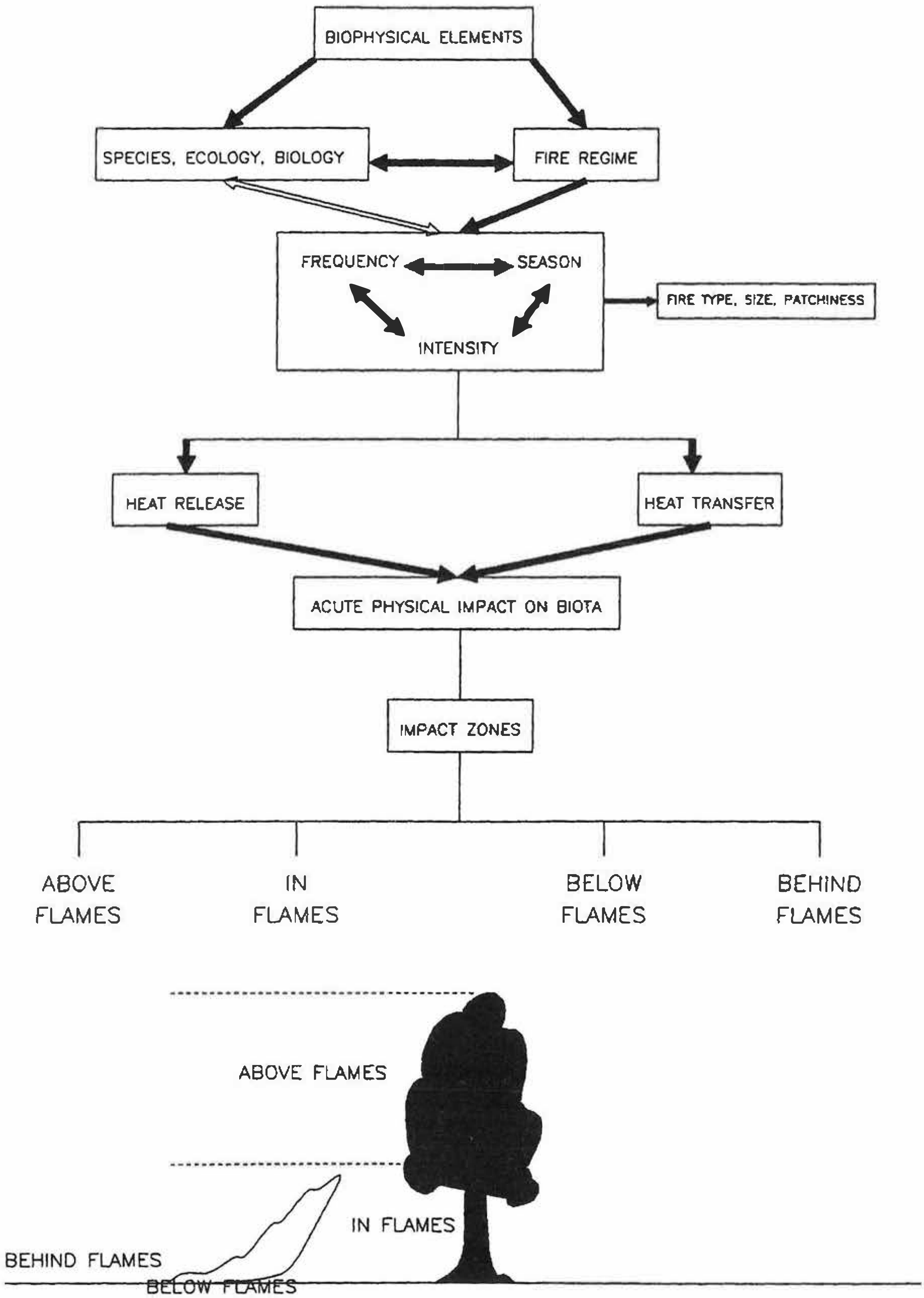


Figure 2. Fire impact linkages

(iii) identify factors that are likely to affect heat transfer to plants and soil; and
(iv) seek correlations between 'easily measured' fire variables, variables which are likely to affect heat transfer, and the impacts.

To study acute impacts of fire, it is convenient to recognize four strata or 'impact zones', as shown in Figure 2 and described in Table 1.

Impact Zone Above the Flames

This zone is affected by hot (but not burning) turbulent gases rising above the flames. The most obvious impact in this zone is crown scorch or leaf browning as leaves, fine twigs and fruits experience a lethal time-temperature regime. For most species in the jarrah forest this leads to crown replacement (Christensen and Kimber 1975; Bell *et al.* 1989) but in other forests (e.g., *Pinus radiata)* plants may be killed outright by full crown scorch. The result is a reduction in cover and density of vegetation and often massive and synchronized seed release.

Flame characteristics, particularly flame height and flame length, are the most meaningful measures for predicting the height at which lethal temperatures (lethal to leaves and fine twigs) occur in this zone because flame size reflects the amount of heat being given off (fire intensity) (Luke and McArthur 1978). An empirically derived relationship between flame height and crown scorch height for jarrah forest fuels is shown in Figure 4 (from Burrows 1994).

The amount of heat transferred to vegetation above the flames is also affected by ambient temperature and wind speed. Semi-empirical models for predicting scorch height (e.g., Van Wagner 1973; Cheney *et al.* 1992; Tolhurst *et al.* 1992) incorporate wind speed and ambient temperature as factors which effect scorch height independent of the effects of these factors on fire behaviour. Van Wagner's (1973) scorch height model is an example of determining an acute impact (scorch height) from a measure of heat energy released (Byram's intensity) and factors affecting heat transfer (ambient temperature and wind speed).

Burrows (1994) found that Byram's fire intensity also relates reasonably well to scorch height experienced in jarrah forests (Fig. 3). Theoretically it should, as it characterizes the total heat output from the flames. Alexander (1982) and Cheney (1990) provide details on how to correctly calculate and interpret fire intensity. A limitation with calculating fire intensity is knowing what fuel and how much of it is involved in the flaming combustion zone. This, and the fact that Van Wagner's relationship was developed from experiments in a different fuel and forest type, probably explains the differences between the models graphed in Figure 3. The best approach is to set some rules which apply for a specific fuel type. For example, for jarrah forest litter bed fuels I assume that the quantity (before fire minus after fire) of dead surface fuel <6 mm and live fuel <4 mm (up to flame height) is burnt in the flaming zone. Rough barked trees such as jarrah can become discrete fuel entities (fuel rods), especially under conditions of high fire danger. Flames will spread in bark up the tree bole, releasing heat close to the tree crowns. This contribution cannot be simply added to the intensity of the surface fire for correlating with crown scorch. I have not been able to devise a sensible mechanism which accounts for this other than to observe that it occurs most frequently when the Soil Dryness Index (Mount 1972; Burrows 1987) is high, fuels are dry (<10 per cent moisture content) and winds are stronger than about 10-15 km h^{-1}. I accept it as part of the unexplained variability of the scorch height-intensity relationship shown in Figure 3. The difficulty with calculating intensity probably explains why the various empirically derived relationships between intensity and scorch height have different coefficients (Fig. 3).

The seasonal differences between the scorch height-flame height relationship (Fig. 4) reflects differences in ambient temperature and moisture conditions and differences in the physiological status of the vegetation.

Impact Zone in the Flames

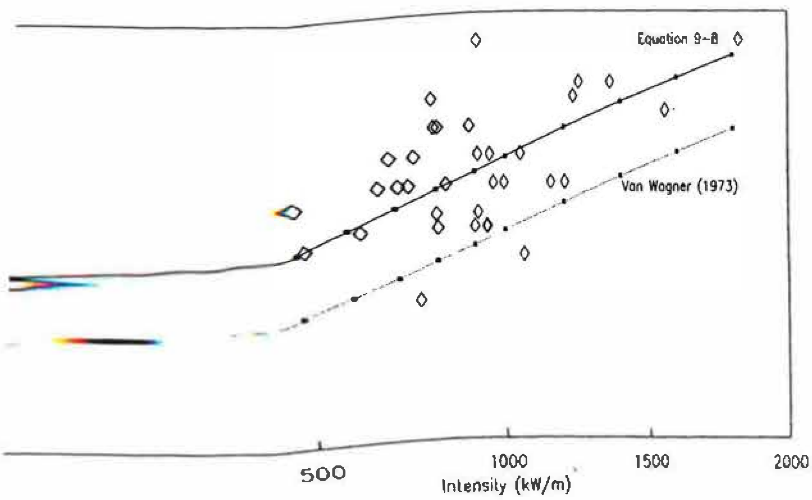
Plants are killed either by defoliation (incineration) or by stem girdling at or near ground level. Defoliation height is about the height of the flames when flame height is less than the height of the vegetation. Most understorey species in the jarrah forest have paper-thin bark and are readily girdled and killed to ground level by even the mildest of fires (most re-sprout from subterranean organs). Trees and tall shrubs which develop thick protective bark are more resistant to death by thermal girdling. Stem mortality will depend on bark thickness, bark moisture content, and the temperatures and duration of heating experienced at the bark surface.

During laboratory fires (Burrows 1994), exposed thermocouple maximum temperatures reached in the flames at a fixed distance (10 cm) above the fuel bed related reasonably well to flame length and fire intensity until a certain flame thickness (emissivity) was reached when maximum temperature saturated. Heat load, or the duration of heating, was independent of intensity but related reasonably well to the quantity of fuel consumed, and to flame residence time. During the same experiments, thermocouples inserted into the cambium of *Banksia grandis* stem sections (near ground level) showed that temperatures experienced during low intensity laboratory fires were largely dependent on bark thickness and on the quantity of fuel consumed and were independent of flame rate of spread.

In contrast to laboratory studies, field studies showed that the mortality rate of *Banksia grandis* and cambial damage to jarrah is affected by bark thickness and fire intensity classes (Burrows 1985). That is, controlling for bark thickness and the quantity of fuel

...ying impacts, descriptors of heat output and factors affecting heat transfer.

	IMPACT ZONES		
...VE FLAMES	IN FLAMES	BELOW FLAMES	BEHIND FLAMES
...rch and death of ...ves, twigs, fruits ...ding to defoliation ...dfall, cover ...luction.	Defoliation, mortaility cambial damage, bark loss, soil exposure, seedbed preparation, seedfall, seed mortaility.	Soil heating, seed germination, soil chemical and strucutral changes, micro-organisms.	Super-heating of soil, allbed, chemical and structural changes, removal/ replacement of hollows, stem damage, tree fall.
...corch height, ...corch area, cover, ...density, quanity of ...seedfall.	Defoliation height, area defoliated, mortality by species, area of cambium damaged, cover, structure, depth of burn quantity of fuel burnt.	Soil temperatures, chemical changes, bulk density, colour soil seed germination.	Area cambial damage, damage, number logs removed, number logs created, trees downed, area of ashbed.
Flame height, flame length, Byram intensity.	Flame height, flame depth, flame residence, time intensity, combustion rate, fuel consumed.	Fuel consumed, fuel size, fuel moisture, flame residence time.	Coarse fuel consumed, consumed, burn out tiem, fuel moisture, drought index.
Temperature, wind speed, height above flames.	Bark thickness, stem size, plant moisture content, bark moisture ambient temperature, drought index.	Soil moisture content, soil bulk density, drought, index.	Soil moisture, soil heat structure, drought index, distance from fuel to tree stem.



...own scorch height as a function of Byram ...tensity for summer/autumn jarrah forest fires.

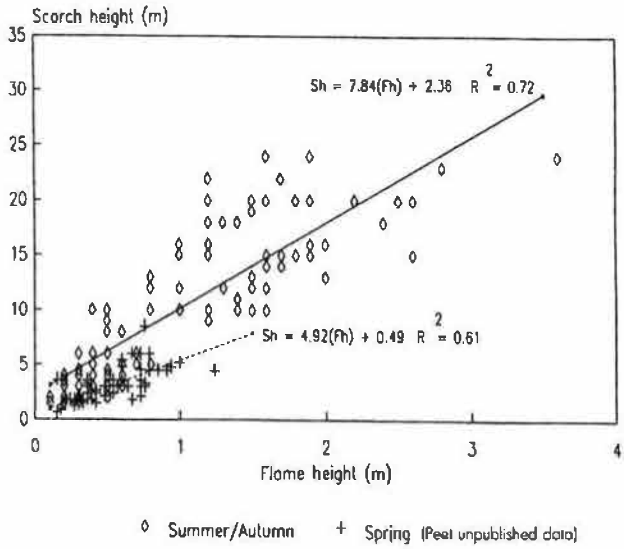


Figure 4. Crown scorch height as a function of flame height for summer/autumn and spring fires in jarrah forests.

consumed, mortality and cambial damage increased with increasing heat flux from the flame, as characterized by intensity classes. It is possible that the highly spiked nature of the heat flux experienced during very intense fires diffuses through the bark to the cambium sufficiently to heat the cells to 60-70°C. An alternative explanation is that in the field, intensity *per se* may not be accounting for increased death and injury, but reflecting burning conditions, especially fuel moisture and wind speed. Fast spreading fires are usually associated with warm, dry and windy conditions and under these conditions, other fuel sources, which may be overlooked, contribute to the heat output of the fire. These include larger diameter surface material and bark on standing trees. Rough barked trees cannot be considered as inert rods or slabs in a fuel bed, but are potential fuel rods in a fuel bed. A significant proportion (up to 30 per cent of total thickness during intense fires) of bark burns on the tree, especially on the leeward side, contributing significantly to cambial damage (Gill 1974; Tunstall *et al.* 1976; Gill *et al.* 1986).

Impact Zone Below the Flames

This zone is the top few centimetres of the soil, including the soil surface. The degree of fire-induced soil heating and subsequent changes to soil chemical and physical properties will depend on how much of the fuel resting on the soil is consumed, how dry the fuel is and the thermal properties of the soil, such as moisture content and bulk density (Aston and Gill 1976; Frandsen and Ryan 1986; Hungerford 1989; Burrows 1994). Soil heating is independent of intensity, although intensity may reflect fuel consumption. The extent to which the soil is heated will affect seed bed preparation, germination of soil stored seed, micro-organism responses and the chemistry and structure of the soil. When fuels and the soil are moist and there is only partial combustion of the litterbed, then top soil is unlikely to be affected. The physical process of heat transfer through soil and some effects of fire on soil are summarized by Aston and Gill (1976), Wells *et al.* (1979), and Humphreys and Craig (1981).

Impact Zone Behind the Flames

This is the zone behind the flaming combustion zone where larger fuel particles such as limbs, logs and old hollow-butt trees burn away slowly, often smouldering rather than flaming. Long durations of localized heating can have considerable impact on the soil and nearby vegetation, giving rise to the 'ashbed' effect, and to severe cambial injury to trees. Hollow-butt trees may burn down. The extent to which this occurs will depend on the amount and distribution of coarse fuel, fuel dryness and the intensity of the fire carried in fine fuels.

CONCLUSION

Fire impacts which are likely to be linked with ecological effects, fire descriptors which relate to these impacts, and factors influencing the heat transfer process are summarized in Table 1 for each of the four impact zones.

I have presented a framework which may be useful for gathering information about fires for ecological study. This framework is comprised of four elements:

(i) biophysical, biological and ecological information;
(ii) fire regime information;
(iii) combustion and heat transfer;
(iv) physical impacts.

Fire will impact directly on plants and animals and indirectly on animals by its impact on plants (habitat).

Fundamental processes of combustion and heat transfer are poorly understood, so local, empirical or semi-empirical models will need to be derived which link acute impacts with meaningful descriptions of heat energy output and distribution.

ACKNOWLEDGEMENTS

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TABLE 1

Impact zone, impacts, quantifying impacts, descriptors of heat output and factors affecting heat transfer.

	IMPACT ZONES			
	ABOVE FLAMES	IN FLAMES	BELOW FLAMES	BEHIND FLAMES
Impacts	Scorch and death of leaves, twigs, fruits leading to defoliation seedfall, cover reduction.	Defoliation, mortaility cambial damage, bark loss, soil exposure, seedbed preparation, seedfall, seed mortaility.	Soil heating, seed germination, soil chemical and strucutral changes, micro-organisms.	Super-heating of soil, allbed, chemical and structural changes, removal/ replacement of hollows, stem damage, tree fall.
Measure of impacts	Scorch height, scorch area, cover, density, quanity of seedfall.	Defoliation height, area defoliated, mortality by species, area of cambium damaged, cover, structure, depth of burn quantity of fuel burnt.	Soil temperatures, chemical changes, bulk density, colour soil seed germination.	Area cambial damage, damage, number logs removed, number logs created, trees downed, area of ashbed.
Descriptors of heat output	Flame height, flame length, Byram intensity.	Flame height, flame depth, flame residence, time intensity, combustion rate, fuel consumed.	Fuel consumed, fuel size, fuel moisture, flame residence time.	Coarse fuel consumed, consumed, burn out tiem, fuel moisture, drought index.
Factors affecting transfer	Temperature, wind speed, height above flames.	Bark thickness, stem size, plant moisture content, bark moisture ambient temperature, drought index.	Soil moisture content, soil bulk density, drought, index.	Soil moisture, soil heat structure, drought index, distance from fuel to tree stem.

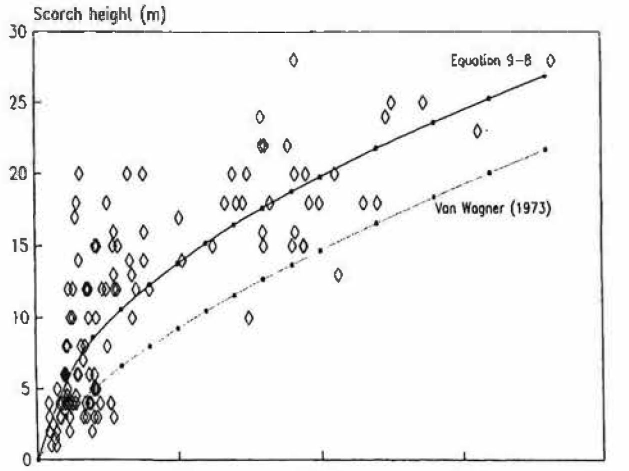
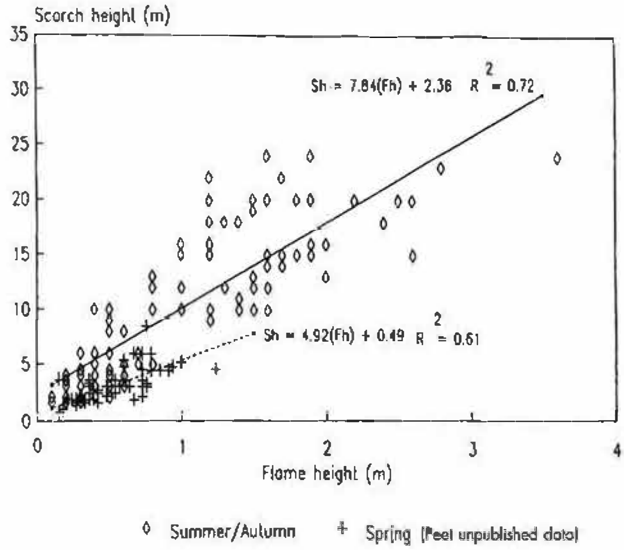


Figure 3. Crown scorch height as a function of Byram (1959) fire intensity for summer/autumn jarrah forest fires.



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Measurement and effects of fire heterogeneity in south-west Australian wheatbelt vegetation

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ABSTRACT

Shrub vegetation in the Western Australian wheatbelt is very spatially heterogeneous, with variation in structure, floristics and litter cover. Using Thermocolor pyrometers and thermocouples, we show that this vegetation heterogeneity translates into heterogeneity in fire-fuel configurations which result in marked variability of fire treatment within individual fires. Vegetation heterogeneity combines with variations in windspeed to produce variation in fire severity at broad and fine scales. Temperatures can vary greatly in the shrub canopy and at and under the soil surface over very short distances and between species. Variation in litter cover also affects temperatures reached, and we show that this is related to variation in the post-fire establishment of seedlings. Fire heterogeneity may be an important factor contributing to the coexistence of shrub species in species-rich heath communities. We argue that some measure of fire heterogeneity needs to be incorporated into fire studies.

INTRODUCTION

Most natural vegetation exhibits a degree of spatial heterogeneity in composition and structure. The scale and degree of heterogeneity varies between vegetation types, with some types exhibiting little heterogeneity (e.g. shrublands with monospecific dominance) and others considerably more (e.g. species-rich heathlands, which exhibit marked structural and floristic variation, or open woodlands which exhibit considerable structural variation).

We contend that this spatial heterogeneity in vegetation results in heterogeneity in the spatial configuration of standing fuel and litter which in turn affects the spatial pattern of fire severity. Our previous studies of the spatial variability of fires in south-west Western Australian vegetation supports this idea, with marked differences in temperatures reached both between vegetation types with different fuel configurations, and between different vegetation components within particular vegetation types (Hobbs and Atkins 1988). This variation in fire treatment may then have important effects on the post-fire vegetation response since fire-induced seed mortality and/or germination stimulation will also vary spatially.

In this paper, we discuss the measurement and effects of fire heterogeneity with particular reference to management fires in heath and shrubland vegetation in the Western Australian wheatbelt.

METHODS

Study Sites

In this paper we discuss two management fires conducted in Durokoppin Nature Reserve, north of Kellerberrin in the central wheatbelt (117° 42' E, 31° 24' S). The vegetation of the reserve is a mosaic of heathland, shrubland and woodland type (Hobbs *et al.* 1989). A 40 ha area in the extreme north-west corner of the reserve was burned on 6 April 1988. Most of the burned area consisted of dense shrubland dominated by *Allocasuarina campestris, Melaleuca uncinata, Calothamnus gilesii* and *Acacia* spp. Shrub heights ranged between 1 and 2.5 m, often with an understorey of smaller shrubs and sedges, especially *Ecdeiocolea monostachya*. Some patches of taller shrubs (3-4 m), woodland dominated by *Acacia stereophylla* (4-5 m), and open heath (0.5-1.5 m) were also present.

The second fire area was 80 ha in the eastern section of the reserve, consisting mainly of open low heath with a high diversity of shrub species varying in height between 0.5 and 2.5 m, with occasional emergents of *Grevillea eriostachya* and *Eucalyptus burracoppinensis*, and areas dominated by *Xylomelum angustifolium, Allocasuarina huegeliana* and *Leptospermum erubescens*. This area was burned on 15 March 1989.

Prefire vegetation sampling was carried out in both fire areas during the spring prior to the fire. Quadrats of 10 m x 10 m were set up throughout the fire areas (18 in the 1988 fire and 20 in the 1989 fire area) and analysed floristically (species complement and percentage cover), and structure was analysed by counting the number of hits by vegetation on 10 levee rods randomly placed in each quadrat. In addition four 5 x 5 m quadrats within the 1989 fire area were harvested for above-ground biomass estimates. Samples were divided into the following fuel components: live stems <6 mm diameter, live stems >6 mm diameter, litter <1 cm diameter, litter >1 cm diameter, and standing dead. Each component was oven dried and weighed.

Fire Temperature Measurements

Two methods of assessing fire temperatures were used. Firstly, thermocolor pyrometers were used, as described by Hobbs *et al.* (1984) and Hobbs and Atkins (1988). These consisted of strips of mica sheet 50 x 37.5 x 0.08 mm, on which stripes of heat sensitive Thermocolor paints were painted. Ten paints were used, each of which changed colour irreversibly at a different temperature over the range 65-770°C. These were painted in an array on two mica strips, which were then joined with paper clips with the paints on the inside surfaces. These pyrometers provide a relatively inexpensive way of obtaining comparative temperature estimates from a wide range of locations within a fire area. In the experimental fires discussed here, pyrometers were distributed over numerous locations, including within the canopies of particular shrub species, on the ground surface, and at 2 cm depth in the soil. Generally 10 pyrometers were placed within each location type.

Following the fire, the pyrometers were retrieved and changes in paint colour recorded. For each location type, a frequency distribution of temperatures experienced was produced. In such diagrams, a uniform, intense fire will produce a straight line, while a heterogeneous fire will yield a more ragged curve (Fig. 1).

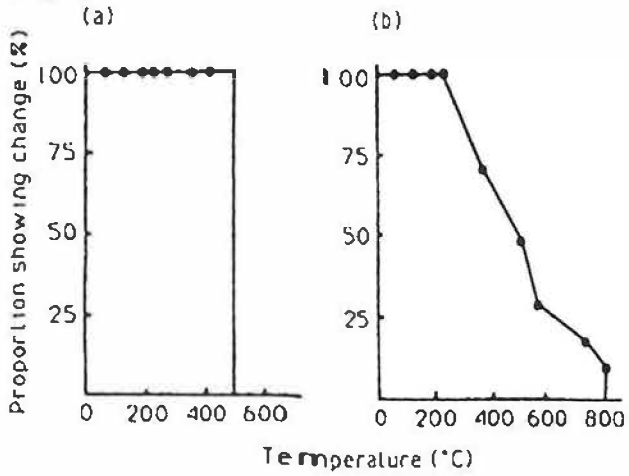

Figure 1. Method of presenting pyrometer results, showing the shape of a graph for hypothetical fires with the same mean temperature (500 °C) which are (a) uniform and (b) heterogeneous (from Hobbs and Atkins 1988).

The second temperature measurement system used consisted of chromel-alumel thermocouples connected via insulated leads to mechanical data loggers. Two loggers, each with 8 thermocouples, were used in each fire. The Grant Instruments ROK8/1r recorders have a 30 s recording interval, and were positioned in fire-resistent vacuum containers, buried in the ground. Thermocouples were placed in a variety of locations, as for the Thermocolor pyrometers, although replication was limited due to the low number of thermocouples available. The thermocouples provided data on the time course of temperature changes during the fire.

Effects of Litter Cover

A feature of the shrub communities studied is the patchy distribution of litter on the ground. Litter accumulates in soil depressions, under shrubs and in other places where lateral movement is restricted. Other areas have predominantly bare mineral soil. We investigated the effects of this heterogeneity in litter cover on fire temperatures and post-fire vegetation regeneration. Prior to the 1989 fire, an area of open heath (approximately 50 x 50 m) was categorized into areas with and without a pronounced litter cover. Ten 50 x 50 cm quadrats were marked in areas with litter, and adjacent to each of these an equivalent quadrat was marked in an area without litter cover. Close to each quadrat, all the litter within an equivalent 50 x 50 cm area was collected, oven dried and weighed. A Thermocolor pyrometer was placed within each marked quadrat at the soil surface (i.e. under the litter where present). In the year following the fire, seedling establishment was followed in each quadrat.

In order to explore further the influence of litter cover on fire temperatures and subsequent vegetation development, we experimentally added litter to plots within the area to be burned. Within the same general area as the 50 x 50 cm quadrats were marked, six 5 x 5 m plots were marked prior to the fire. In three of these, dry foliage and stem material of *Allocasuarina campestris* harvested from nearby was placed on the ground so that bare soil areas were covered in this simulated 'litter' at a rate of approximately 500 g^{-2} (5 t ha^{-1}). Adjacent plots were left with no added litter. Thermocolor pyrometers were placed at 1 m above ground, the soil surface and 2 cm soil depth within each of these sets of plots. In May 1990, 14 months after the fire, counts of seedlings were taken in ten 50 x 50 cm quadrats randomly placed within the litter added plots and an equivalent set in the plots without litter.

RESULTS AND DISCUSSION

Vegetation and Fire Heterogeneity

The vegetation in both areas was very heterogeneous, with marked spatial variation in composition and structure (Figs 2, 3). This variability was apparent both at a broad scale (e.g. between quadrats in different locations separated by hundreds of metres in the burn areas) (Fig. 2a and b), and at a narrower scale, such as within individual 10 x 10 m quadrats (Fig. 2c) or between quadrats located 5-10 m from each other (Fig. 3). Our studies have also indicated that considerable floristic differences can occur between closely located quadrats (R.J. Hobbs and L. Atkins, unpublished data).

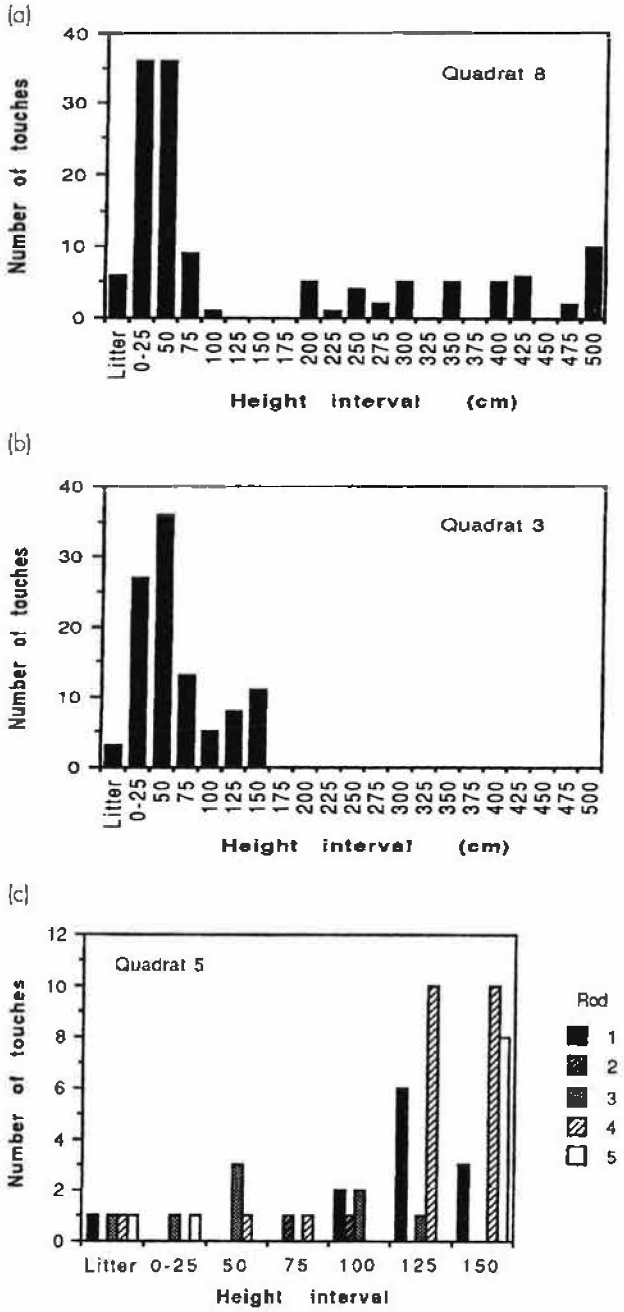

Figure 2. Results from levee rod sampling within 10 x 10 m quadrats in shrub vegetation in western Durokoppin Nature Reserve prior to burning. Graphs indicate the total number of touches by vegetation at height intervals from ground level to 5 m on 10 rods in each quadrat (Quadrats 8 and 3), and the number of touches on five individual rods within one quadrat (Quadrat 5).

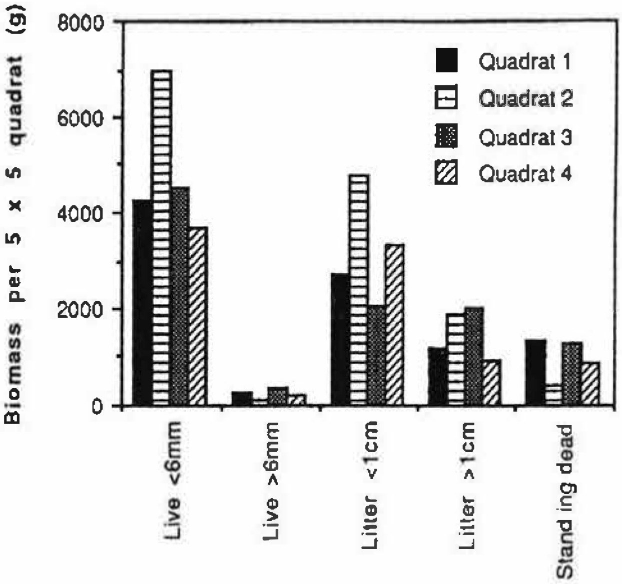

Figure 3. Above ground biomass (oven dry-weight) of various components harvested from four 5 x 5 m quadrats taken in close proximity in sandplain heath prior to fire in east Durokoppin Nature Reserve.

The 1988 fire resulted in a very patchy burn (Fig. 4), with a range of fire severities over the burn area. Two factors appear to have contributed to the patchiness of the fire, i.e. vegetation patchiness and variations in windspeed. Windspeeds varied considerably during the fire ranging from 0-15 km h^{-1}. The fire report prepared by K.J. Atkins (CALM) reads:

> 'The fire moved rapidly into the surrounding bush until the windspeed dropped, then the fire suddenly slackened. If there was no ground fuel, such as a *Borya* herbfield, then the fire would tend to be dampened right off with occasional flareups in areas of denser vegetation. An increase in windspeed may cause an increase in fire behaviour, but often the fire had dropped to such an extent that there was no such response. .. ..In areas where there was a ground fuel present the fire would display more intense behaviour and with a drop in windspeed would still burn as a ground fire. Increases in windspeed would result in a rekindling of fire behaviour.'

Thus, variation in fire severity was caused by two interacting factors. Dense vegetation and ground cover led to a more intense and/or continuous burn, especially at higher windspeeds. Less dense vegetation still ignited, but generally only at higher windspeeds. The interaction of vegetation heterogeneity and variation in windspeed is thus likely to produce a very variable fire treatment. A further variable which we did not include in this study was canopy moisture content, which might also vary spatially and between species.

The 1989 fire was conducted with more uniform and stronger winds (10-20 km h^{-1}) and resulted in a considerably more complete burn. Even so, unburned and partially burned patches remained, presumably as a result of the same interplay between fuel distributions and windspeeds.

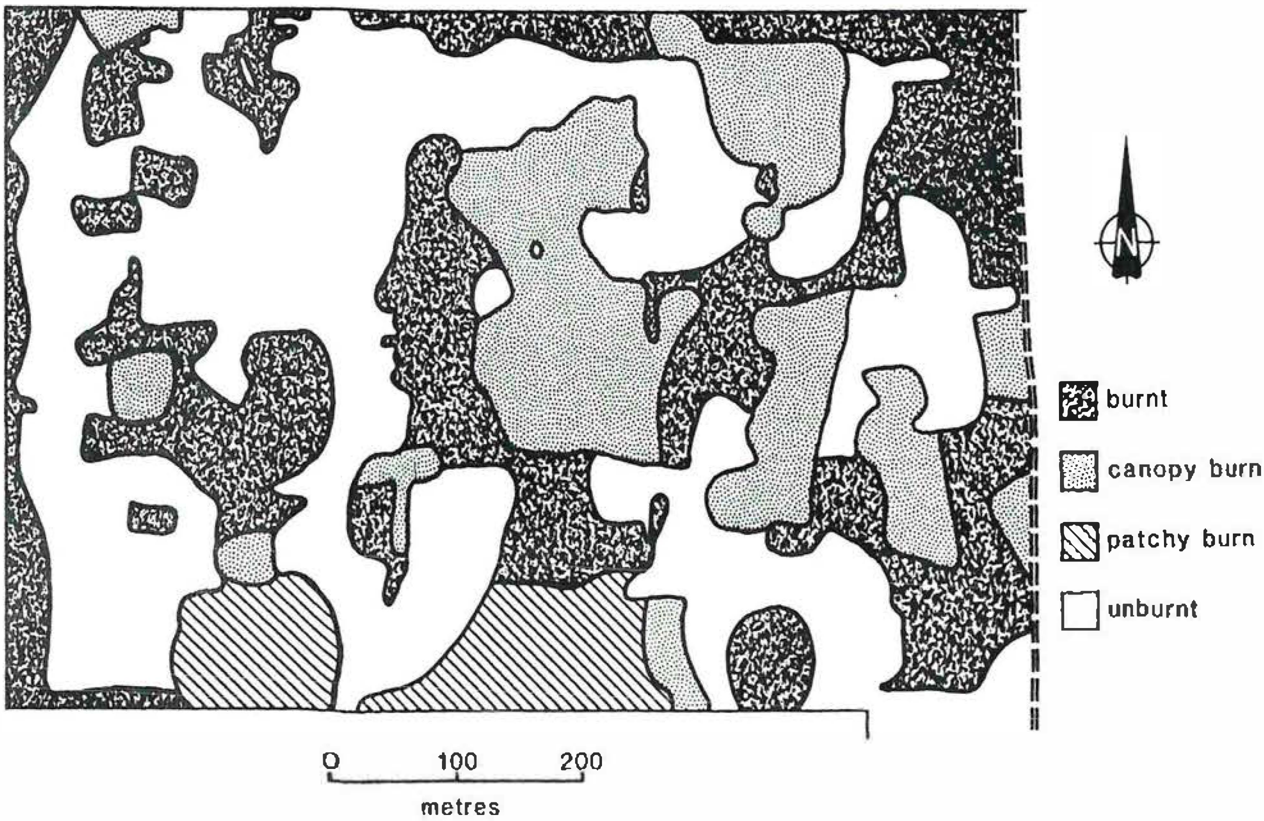

Figure 4. North-western portion of Durokoppin Nature Reserve following management fire in April 1988, indicating variation in severity of fire. Burnt = complete burn of understorey and ground layer, without complete removal of canopy layer; Canopy burn = complete removal of canopy and understorey; Patchy burn = partial burn with some unburnt areas; Unburnt = no fire treatment.

Temperature recordings during the two fires indicate that the scales of variation found in the vegetation are replicated in the range of temperatures experienced. Thus, within marked 10 x 10 m quadrats, considerable variation occurred in temperatures experienced in the canopy and at and below the soil surface (Fig. 5). Thermocouples placed in and under different plant species situated within metres of each other indicate that quite different fire treatments can be experienced by plants within close proximity, even in what was considered a reasonably intense fire (Fig. 6). As well as this fine scale variation in temperatures, larger scale differences can also be observed. Data from three 10 x 10 m areas situated about 10-20 m apart (Fig. 7) indicate that areas with approximately similar vegetation (i.e. mixed heath) can have markedly different fire treatments, with no soil heating in one case and soil heating to 500°C in another. Variation is also caused by the presence of particular species, such as the emergent *Eucalyptus burracoppinensis*. In this case, a considerably more uniform fire treatment was experienced, with soil temperatures reaching 770°C, presumably as a result of an increased litter cover.

Addition of litter to mixed heath quadrats altered the fire treatment markedly, with higher, more uniform temperatures experienced in the canopy and at and below the soil surface following litter addition (Fig. 8). This is reflected in data from marked areas with and without litter before the fire, and temperatures reached at the soil surface were significantly higher where litter was present (Table 1). Bradstock *et al.* (1992) have also discussed the influence of litter type and depth on soil temperatures in mallee shrublands.

Litter and Post-fire Vegetation Response

Data from marked quadrats with and without litter prior to the fire indicate significant differences in seedling establishment between the two microhabitats. Several species had significantly more seedlings in the litter areas, while others had more in the areas with no litter (Table 1). There was no apparent relationship between seedling numbers and species of adult shrub present before fire. One might expect greater seedling establishment in areas with litter since seeds are liable to collect in these areas and there may be some response to post-fire nutrient release in ash. Ash deposits were, however, dispersed by wind in the days following the fire. The fact that some species were more abundant in areas without litter indicates that these factors alone do not account for the post-fire distributions of seedlings. Data from quadrats where litter was experimentally added support this contention (Table 2). Here, the same species which were abundant in the areas with litter occurring naturally (Table 1) were also abundant

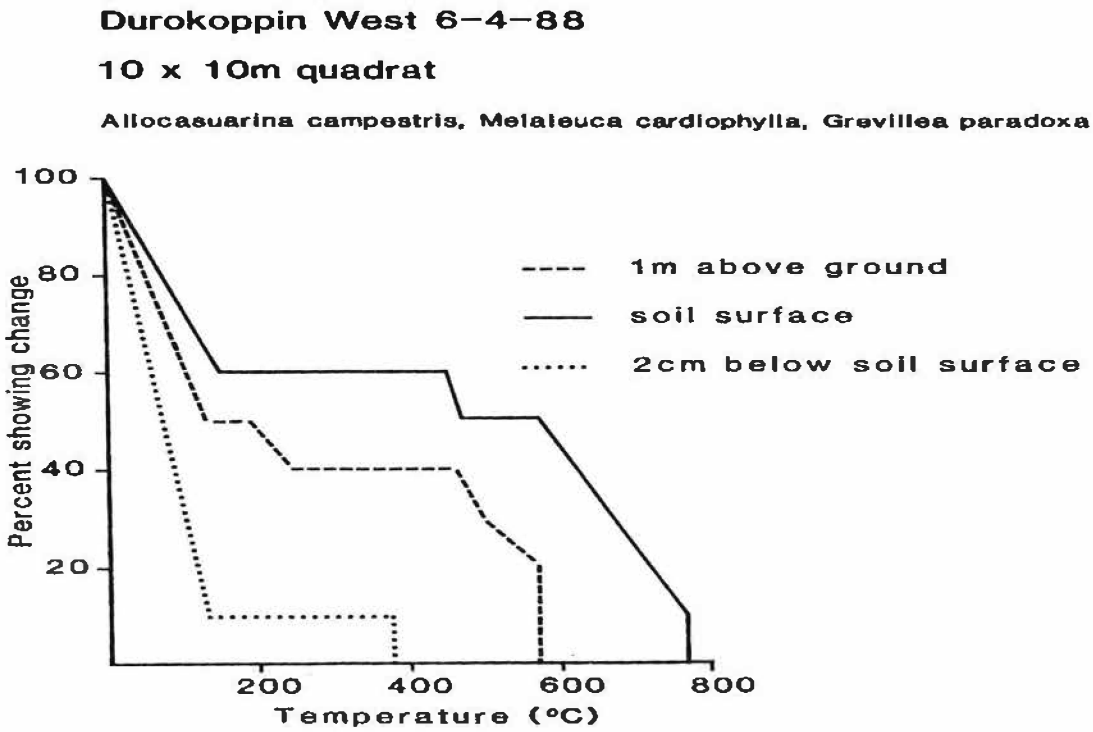

Figure 5. Thermocolor pyrometer results from within one 10 x 10 m quadrat during the fire in western Durokoppin in April 1988. Results from ten pyrometers in each location.

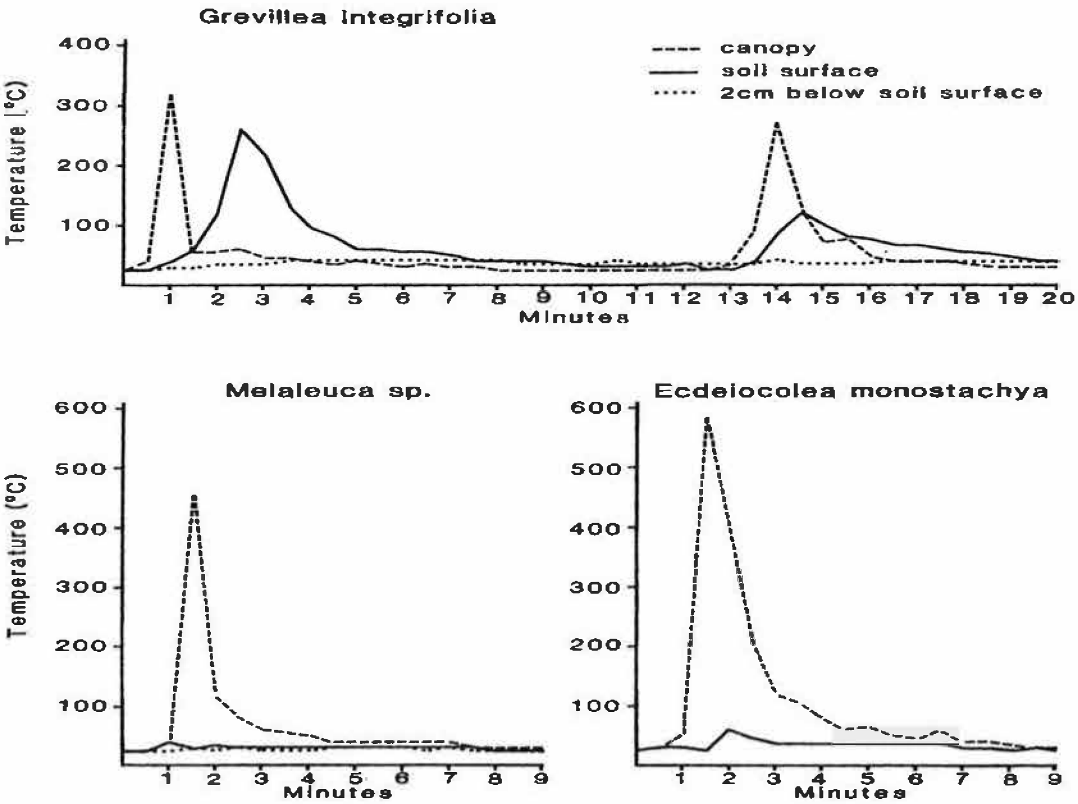

Figure 6. Results from thermocouples located in and below the canopies of three different species within a 10 x 10 m quadrat in sandplain heath burned in March 1989. The double peak in Grevillea integrifolia *represents the passage of the initial fire front and a second front which spread laterally some time later, and illustrates the potential complexity of fire treatment within heterogeneous fuels.*

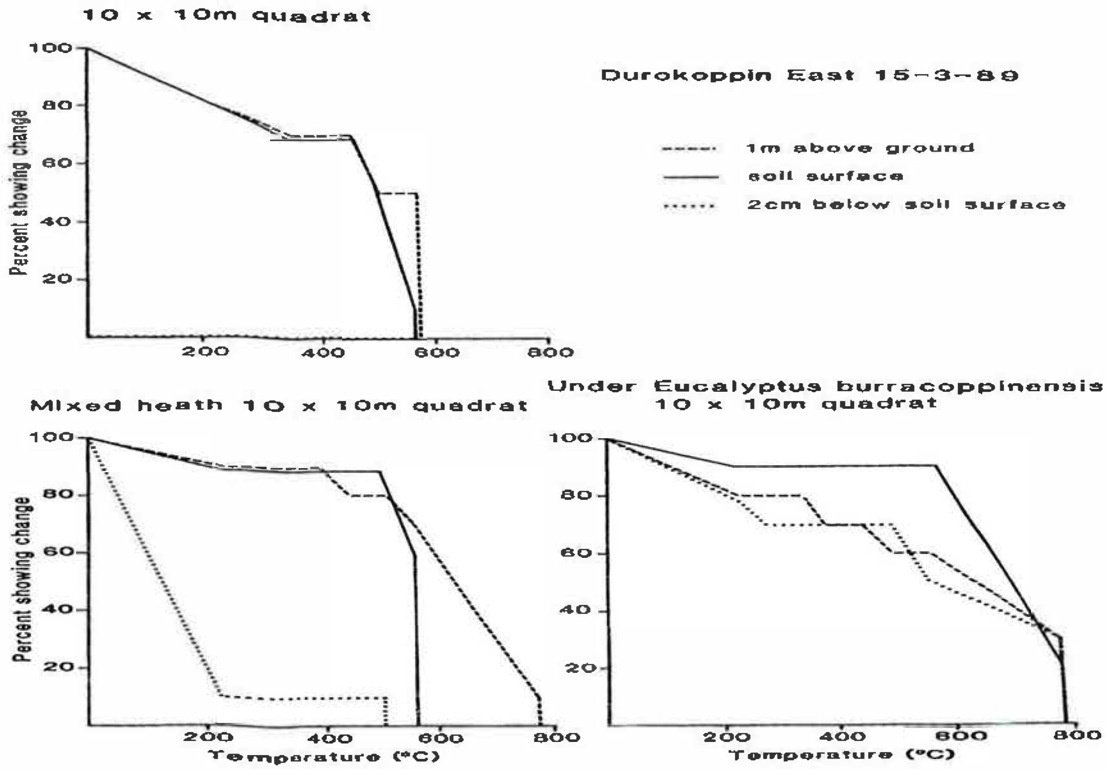

Figure 7. Thermocolor pyrometer results from within three different closely located 10 x 10 m quadrats in sandplain heath burned in March 1989. Two quadrats were in mixed heath and the third was located within an individual of Eucalyptus burracoppinensis. *Results from ten pyrometers in each location.*

TABLE 1

Dry weight of litter present before fire, maximum temperature recorded at the soil surface during fire, and numbers of seedlings of most common plant species present 14 months after the fire in marked areas with and without litter cover in kwongan in east Durokoppin Nature Reserve, 1989. Data from ten 50 cm x 50 cm quadrats in each microhabitat type (mean ± 1 S.E.), with significant differences between microhabitats indicated, as determined by t test for litter weight and temperature, and Mann-Whitney U test for seedling numbers. Litter was collected from 50 x 50 cm quadrats located near to marked areas immediately prior to the fire in March 1989.

	LITTER		BARE SOIL
Litter dry weight (g m^{-2})	212 ± 24	$p < 0.001$	8 ± 3
Maximum temperature, soil surface	665 ± 36	$p < 0.01$	560 ± 0
NUMBERS OF SEEDLINGS			
Allocasuarina acutivalvis	3.1 ± 0.8	$p<0.05$	0.5 ± 0.2
Allocasuarina campestris	1.9 ± 0.9	$p<0.05$	0.2 ± 0.2
Baeckea floribunda	2.4 ± 1.6	-	0.3 ± 0.3
Grevillea integrifolia	0.6 ± 0.4	-	0.3 ± 0.1
Grevillea pritzelii	1.3 ± 0.5	$p<0.05$	0.3 ± 0.2
Leucopogon hamulosus	1.0 ± 0.5	$p<0.05$	20.1 ± 10.7
Verticordia chrysantha	0.4 ± 0.2	$p<0.05$	1.3 ± 0.3

TABLE 2

Numbers of seedlings of common species present in May 1990 within 5 x 5 m plots which received additional litter (approximately 500 g m^{-2}) prior to fire in March 1989 and adjacent areas with no added litter, within kwongan vegetation in east Durokoppin Nature Reserve. Data are from ten 50 cm x 50 cm quadrats taken randomly within 3 litter-added plots and 10 taken in immediately adjacent 5 x 5 m areas (Mean ± 1 S.E., with significance of difference indicated, as determined by Mann-Whitney U test).

	LITTER ADDED		ADJACENT
Acacia nigripilosa	3.5 ± 1.2	$p< 0.05$	0
Allocasuarina campestris	3.9 ± 1.4	$p< 0.05$	0.6 ± 0.6
Grevillea integrifolia	2.2 ± 1.5	-	5.4 ± 2.0
Grevillea pritzelii	5.7 ± 2.2	$p< 0.05$	0.8 ± 0.4
Leucopogon hamulosus	1.8 ± 1.1	$p< 0.05$	17.0 ± 5.0
Verticordia chrysantha	0.2 ± 0.2	$p< 0.05$	2.2 ± 0.6

in areas with litter added. Similarly, species most abundant in areas with no litter were also more abundant in the plots with no addition of litter. If seedling distributions were related only to the pre-fire distribution of litter and its effect in trapping seed, addition of litter shortly before the fire should have had no impact on seedling distributions. All the species recorded here store seed in the soil, except the two *Allocasuarina* species, which maintain canopy seed stores. However, the fact that seedling numbers of *Allocasuarina* spp. were higher in the quadrats with litter and in quadrats with litter added indicates either that these species may also have a soil seed store with heat-stimulated germination, or that seed dispersed from the canopy germinates preferentially (or seedlings survive better) in areas with a pre-fire litter cover. Clearly, more detailed study is required to investigate these possibilities.

The results suggest that the effect of litter on fire treatment may be an important factor in determining post-fire seedling distributions. The mechanism for this is likely to be differential responses by the seeds of different species to fire treatment. It is likely that seeds of some species require a high temperature treatment to stimulate germination (Cushwa *et al.* 1968; Martin *et al.* 1975; Portlock *et al.* 1990; Auld and O'Connell 1991; Bell *et al.* 1993), while seeds of other species will be intolerant of high temperatures. Differential heating

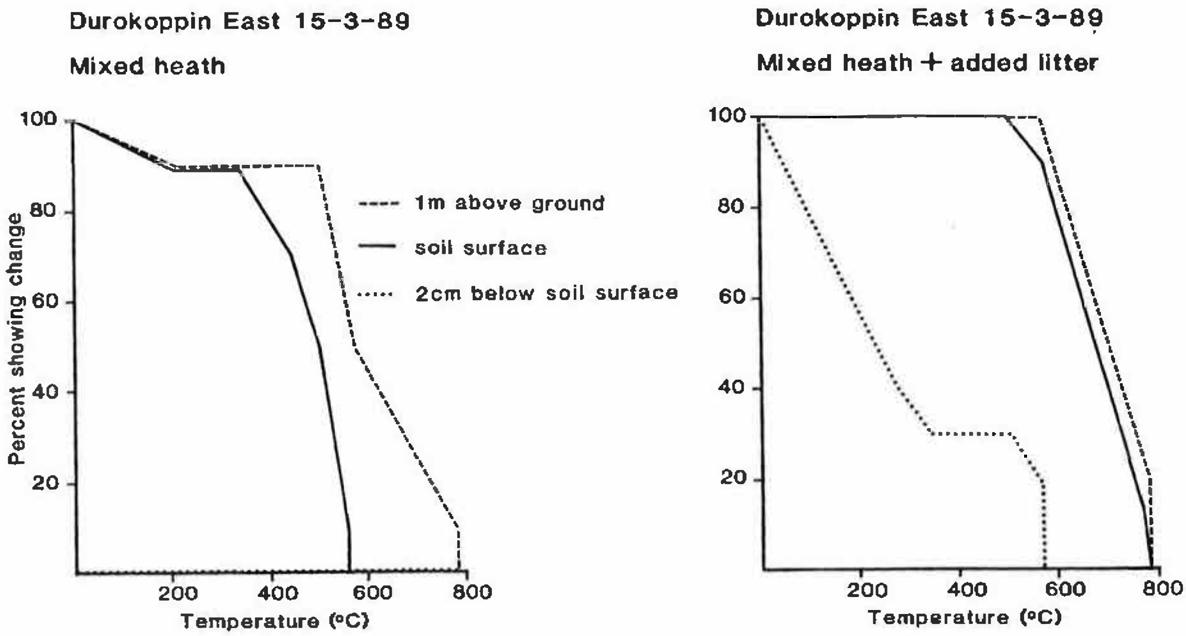

Figure 8. Thermocolor pyrometer results from 5 x 5 m quadrats in sandplain heath burned in March 1989, with and without the addition of litter. Results from a total of 10 pyrometers placed within quadrats of each type.

of seeds stored either in the canopy or in the soil will then result in a heterogeneous response to fire. We do not have data on the effects of elevated temperatures on germination of the species found in our study quadrats, but data on understorey species from *Eucalyptus salmonophloia* woodland, another major vegetation type in the region, indicate that species' responses can vary markedly (Fig. 9). Some species were inhibited and others stimulated by high temperatures, and subtle differences in temperature response are evident. Species coexistence may thus be facilitated by their differential responses to a heterogeneous fire treatment.

In this respect, fire heterogeneity may be an important mechanism promoting the maintenance of species diversity in shrublands and other vegetation types, in addition to other mechanisms proposed (Hnatiuk and Hopkins 1980; Hopkins and Griffin 1984). Other studies have indicated the importance of heterogeneity in post-fire litter distribution in determining species' establishment success (Enright and Lamont 1989; Lamont *et al.* 1993), but the present results indicate that pre-fire distributions can also be important.

CONCLUSIONS

In this paper we have indicated that heterogeneity in fire severity is an important aspect of fires in shrub communities. This heterogeneity can be attributed to variability in ground and canopy fuel configurations, possibly in conjunction with variation in wind strength during the course of a fire. The fires studied were both controlled management fires, and it is possible that such heterogeneity would be less evident in a wildfire situation. Nevertheless, even under such conditions it appears likely that variations in windspeed and in the distribution of canopy and, especially, litter fuels would induce some spatial variation in fire treatment. This spatial variation in fire severity in turn influences the patterns of post-fire regeneration.

Is this spatial variability in fire severity important? Certainly, in order to derive an understanding of the processes influencing vegetation response to fire, it is important to recognize that such variability exists (see also Gill 1981). Fire studies frequently rely on simple descriptors which summarize the fire treatment, using mean fuel loadings and single parameter intensity measures. While this is probably appropriate for broad management considerations, for more focussed analysis of fire effects more consideration of fire heterogeneity is needed. The biota exhibits a varied response to fire, and heterogeneity in fire severity clearly has important ramifications for the long-term persistence of that biota. For management, the implication seems to be that heterogeneity of fire treatment should be aimed at; uniform low or high temperatures will not provide the range of treatments and hence cues for the range of species present.

The measurement of fire heterogeneity is an intensive process, requiring adequate replication and spatial coverage. The pyrometers more efficiently captured the range of spatial variation in temperature than did the thermocouples, since these were restricted in number. Unfortunately, Thermocolor paints are no longer available, and tests of alternatives have shown them to be less satisfactory. However, since variation in fire severity can be linked directly to fuel configuration and is also probably mediated by windspeed variations, it should be possible to derive estimates of fire heterogeneity from these two variables. This then involves building variation into standard methods of estimating fire intensity.

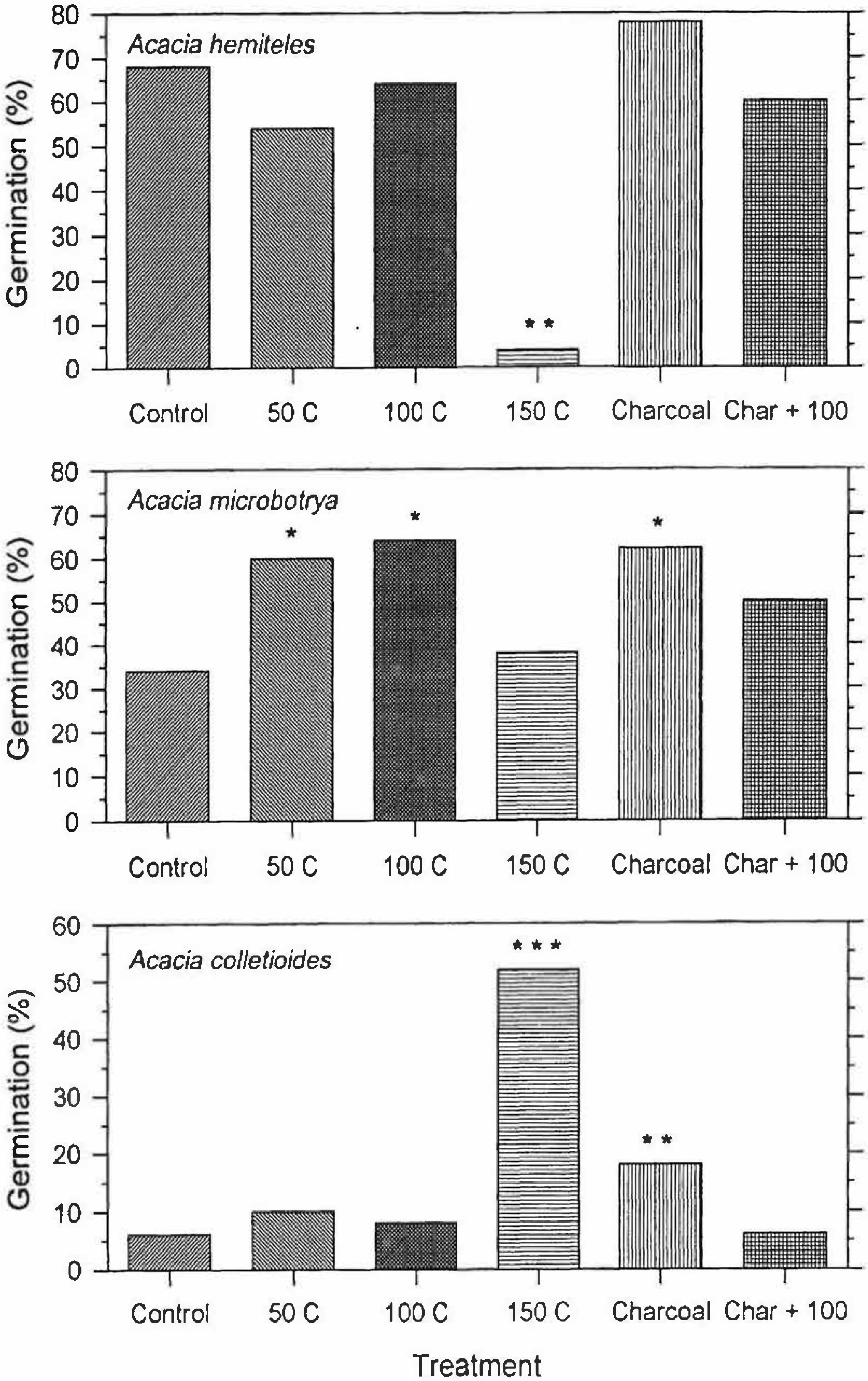

Figure 9. Germination (percentage) of three Acacia *species which are common components of the understorey in* Eucalyptus salmonophloia *woodland in the central wheatbelt, following various treatments. Seeds were scarified and subjected to different temperatures in an oven for 10 minutes, placed on filter paper in petri dishes and watered regularly for 38 days. In addition, charcoal fragments were added to some dishes. Controls received no heat or charcoal. Five dishes were assigned to each treatment, with ten seeds per dish. Asterisks indicate significant differences from control, as determined by t test on square root transformed data: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.*

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Tree mortality in relation to fire intensity in a tropical savanna of the Kakadu region, Northern Territory, Australia

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ABSTRACT

Frequent fire is a feature of the tropical savannas of the Kakadu region of the Northern Territory. In a landscape-scale experiment at Kapalga Research Station in Kakadu, fires have been lit experimentally in catchments 15-20 km² in area, both early in the dry season (June) and late in the dry season (September) between 1990 and 1993. In ***Eucalyptus miniata*** - *E. tetrodonta* forest the mean intensity of early dry season fires (4200 kW m^{-1}) was less than that of the late dry season fires (7800 kW m^{-1}), although there was considerable spatial and yearly variation. In 1991-92, two years after the commencement of the fires, tree mortality was greatest in compartments burnt late in the dry season (15.4 per cent), intermediate in compartments burnt early in the dry season (7.6 per cent) and least in the unburnt compartments (3.5 per cent). Season of burn, however, was not a statistically significant predictor of tree mortality; maximum fire intensity (fitted as a co-variate) explained the majority of the variation in tree mortality. Measurement of fire intensity, which can be used as a co-variate in experimental fire-effects studies, is thus a vital component of fire experiments where season of burn is a factor within the design.

INTRODUCTION

Frequent fire is a feature of the tropical savanna forests and woodlands of the 'Top End' of the Northern Territory. Tens of thousands of square kilometres burn each year during the dry season (Press 1988; Graetz *et al.* 1992), and the vast majority of fires are deliberately lit by humans. The vegetation of the region is predominately open savanna, with a discontinuous overstorey of ***Eucalyptus*** spp. and a more continuous understorey of annual and perennial grasses, forbs and small shrubs (Wilson *et al.* 1990). Press (1988) estimated that in the Alligator and Adelaide Rivers region, over 50 per cent of all lowland forest systems in all land use categories carried fire during the 1980-1985 period. The fire-prone nature of the region is a consequence of the strongly seasonal climate. Summer rainfall is relatively high (800-1800 mm) and supports consistent annual growth of the understorey. The dry season is long (6-9 months), hence fuels cure each year. Fire has been used for millennia by local aboriginal people, with a 9.5 month fire season, and a peak of activity in July (Braithwaite 1991). Fire is used currently as a management tool by most land users within the region, and most fires are deliberately lit. Reasons for burning include fuel/hazard reduction, promotion of more palatable herbs and grasses, and emulation of traditional aboriginal burning practices.

Within the World Heritage Kakadu National Park, prescribed fire is used within the savannas and grasslands by the managing agency (Australian National Conservation Agency; Anon. 1989). Burning commences in the early dry season (May), and is mostly complete by August. Braithwaite and Estbergs (1985) estimate that the savannas of Kakadu are burnt two years in every three. Fire, and its effects, have been little-studied in Kakadu. CSIRO is presently conducting a landscape-scale fire experiment in savannas at the Kapalga Research Station within Kakadu National Park. The aim of this experiment is to assess the effects of various fire regimes on the soils, plants, invertebrates and vertebrates of the savannas (see papers by G. Cook, R. Braithwaite, this volume). This paper presents data on fire behaviour with respect to season of burn for the years 1990-1992, and effects of fire on tree mortality.

METHODS

Study Site

Kapalga Research Station is 180 km east of Darwin and occupies some 700 km². Open eucalypt savannas predominate on the well-drained lateritic soils of the drainage divides. Floodplain areas are occupied by treeless grasslands, and there are several pockets of monsoon vine forest. For the fire experiment, the region has been subdivided into a number of management compartments, which represent the catchments of minor streams which drain into the surrounding major river systems - the West Alligator and the South Alligator Rivers. Each compartment is 15-20 km^2 in area. Within each compartment a permanent, reference transect, 700 m long, has been established along a topographic-soil-moisture gradient, from the more poorly-drained shallow sands of the creek margins, to the better drained loamy soils of the drainage divides; relief along the transects is 10-30 m. The vegetation of the creek margins is woodland dominated by *Eucalyptus alba* and *E. papuana*; the understorey is mostly perennial grasses. The vegetation of the better-drained soils is open forest of *E. miniata* and *E. tetrodonta*, with the understorey dominated by a mixture of annual grasses such as *Sorghum intrans* and perennial grasses such as *Heteropogon triticeus*.

Fire Treatments

Each compartment is subject to one of four fire regimes:

(1) 'Early dry season'; burnt once during the early part of the dry season (June).
(2) 'Progressive'; burnt 3 times (June, July and September) during the dry season as the fuels cure; the spatial pattern of burning is concentric, from the outside of the compartment to the inside.
(3) 'Late dry season'; burnt once during the late dry season (September).
(4) 'Natural'; unburnt control; fire is excluded from these compartments, although one or two small scale (< 1 ha) fires have been detected over the study period.

For each of the early, progressive and late fire regimes, there are three replicate compartments; the natural regime has four replicates. Fires are lit from vehicle-based ground crews, and controlled by a series of double-fuel breaks between compartments. The burning regimes commenced in June 1990, and have been applied annually since then. The data presented in this paper are from each of the three early and three late compartments and three of the four natural compartments.

Fire Measurement

Byram fire-line intensity was determined for most of the experimental early and late dry season fires lit between 1990 and 1993. Intensity, I, is a measure of the energy release along the fire front, and is defined as the product of the heat content of the fuel, fuel standing crop, and the rate of forward spread of the fire line or perimeter; the units are kW m^{-1} (Byram 1959). The mineral-free heat of combustion of the fuel was assumed to be 20 000 kJ kg^{-1}. Rate of forward spread was determined using a series of electronic fire timers over a representative 0.5 ha area, as described by Moore and Gill (this volume). Fuel weight (kg m^{-2}) was determined directly by 5 x 0.25 m^2 quadrats. For those compartments where intensity was not measured directly, intensity measurements were based on rates of spread observed by the fire crews, coupled with empirically derived relationships between fire intensity and canopy scorch height (A.M. Gill, P.M. Moore and R.J. Williams; unpublished data). Based on the experience of the fire crews and those involved in direct measurement, these estimates are considered to be accurate to within 1000 kW m^{-1}, or about 10 per cent.

Tree Mortality

The impact of the fires of 1990 on tree mortality was assessed by determining the abundance of standing dead trees along the permanent transects in each of three of the early, late and natural fire compartments. Measurements were restricted to one widespread vegetation type: open *E. miniata-E. tetrodonta* forest on well drained lateritic soils. The survey was carried out at the end of the wet season (April-May) 1992, two full growing seasons after the initial fires. The total number of trees taller than 3 m in a 200 x 20 m section of each permanent transect in this forest type was counted for each of the three replicate compartments of the natural, early and late fire regimes. For each tree, the following attributes were determined: species, x, y co-ordinates, height, diameter at breast height, and life state (based on presence/absence of epicormic or basal resprouts; trunks without either were scored as dead). Approximately 2000 trees in total were sampled.

Data Analyses

Fire intensity data were analysed by repeated-measures ANOVA, following log transformation. Tree mortality was analysed by GLMs, using a binomial error structure and logit link function; the design strata are displayed in Table 1. Fitted terms included fire intensity (fitted as an explanatory variable), season of fire (early, late, natural) and size class of individual (>3-<6 m, >6-<9 m, >9-<12 m, >12-<15 m, >15 m). The effects of both fire intensity and season of fire were tested against the fire x replicate interaction; the effect of size class

TABLE 1

The design strata of the terms and interactions of the model fitted to the tree mortality data.

TERM	DF
Intensity	1
Season of burn	2
Fire.Rep (=Compartment)	5
Height class	4
Intensity x height class	4
Season x height class	8
Residual	20
Total	44

was tested against the overall residual.

The analyses were conditioned by several limitations of the design. First, the assessments of forest condition (canopy cover and standing dead) were undertaken at the end of the second growing season (1991-1992), that is after two years worth of fires - 1990 and 1991. For the analyses of co-variance, the maximum intensity of the fires applied to a given compartment during either the 1990 or 1991 dry seasons was fitted as the co-variate (explanatory variable). The primary reason for this was that for all but two of the compartments the measured intensities between years varied by between 5 and 15 times, with the fires of 1990 being overwhelmingly more intense. For the two compartments wherein this was not the case, fire intensity was similar between years.

Secondly, one compartment in the late dry fire regime (Compartment F) was accidentally burnt in 1990 at the time of the early dry season fires. It was burnt the following year at the experimentally correct time (September), with an intensity very similar to that of the previous year. To allow for this design problem, two sets of analyses of mortality were undertaken. The first set was based on a non-orthogonal design with respect to season of burn, with three replicates for the natural fire treatment, four replicates for the early dry season fire treatment, and two replicates for the late dry season fire treatment. The second set was based on an orthogonal design, with three replicate compartments per fire regime, as if the accidental burning of Compartment F in June 1990 had not occurred.

RESULTS

Fire Intensity

The average fire intensity for the early and late dry season fires for the years 1990-1992 are given in Table 2. The mean, overall intensity of early dry season fires (4200 ± 1000 kW m^{-1}) was approximately half that of the late dry season fires (7800 ± 1750 kW m^{-1}), although there was considerable spatial and yearly variation. The fires of 1990 were the most intense. The peak fire intensity was 18 000 kW m^{-1}, recorded on one of the late-fire compartments (Compartment G) in September 1990. Fires between 7000 and 10 000 kW m^{-1} occurred on the early-fire compartments in June 1990. In 1991, fire intensity was substantially less than that of 1990, in both the early and late fires; there was little difference in the average intensity of the early and late fires during that year. In 1992, the intensity of the early fires was about one-third that of the late fires. Despite the occurrence of several fires of relatively high intensity (>10 000 kW m^{-1}) over this three-year period, crown fires did not occur. Crown scorch, however, was complete in all fires of intensity greater than *ca* 7000 kW m^{-1}.

Tree Mortality

Tree mortality, as expressed by the average percentage of standing dead trees taller than 3 m, in relation to season of fire is given in Table 3. For these data, Compartment F (the 'late' compartment accidentally burnt 'early' in 1990) is included within the late dry season regime; averages determined by including the data from Compartment F within the early dry season fire regime were within 3 per cent of these figures. Mortality was greatest in the late-burnt compartments (15.4 per cent); the comparative figures for the early-burnt and unburnt compartments were 7.6 per cent and 3.5 per cent (Table 3). There was substantial within-treatment variation, however, and the effect of season of fire on mortality was not significant ($p>0.1$; 2,5 Df). Analysis of co-variance, however, with maximum fire intensity over the 1990-1991 period as the explanatory variable, indicated that the effect of fire intensity on tree mortality was significant ($p<0.05$; 1,5 Df); both non-orthogonal and orthogonal ANCOVAs gave similar results.

TABLE 3

Tree mortality, as expressed by the average percentage (± SE) of standing dead trees taller than 3 m which were dead in April 1992, by fire treatment (Natural or unburnt control, Early dry season fire, Late dry season fire). Figures are fitted values, based on GLMs which include fire intensity and tree size as terms, pooling height class.

FIRE TYPE		
Natural	Early Dry season	Late Dry season
3.5 ± 0.9	7.6 ± 3.6	15.4 ± 7.3

TABLE 2

The average fire intensity (kW m-1; ± SE) for the early and late dry season fires for the years 1990-1992, and the average (AVG) pooling years.

	1990	1991	1992	AVG
EARLY	8300 ± 700	3200 ± 1300	1900 ± 500	4200 ± 1000
LATE	13300 ± 1900	3700 ± 2100	6300 ± 1000	7800 ± 1750

Tree mortality was also significantly affected by tree size (Fig. 1) and species (Table 4). Mortality was greatest in the smallest size class (3-6 m) and least in the largest size class (>15 m). Mortality was also affected by species, being less in the canopy dominants - ***Eucalyptus miniata* and *E. tetrodonta*** - than in the canopy subdominants such as the leguminous tree ***Erythrophleum chlorostachys***.

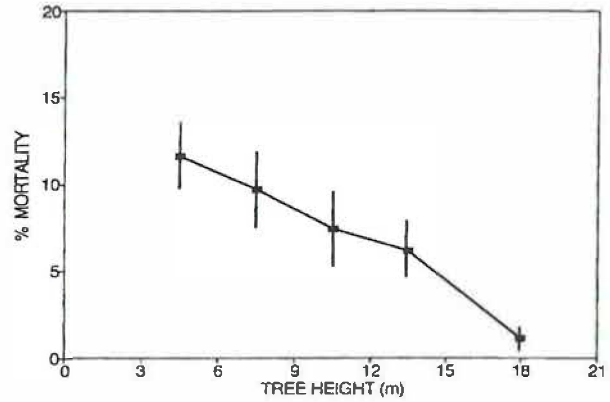

Figure 1. Tree mortality as a function of tree size. Figures are fitted values (+ SE), pooling season of fire and species. Tree heights are the mid-points of the height classes indicated on the X-axis.

TABLE 4
Fitted values of mortality by species, based on GLMs which include fire intensity and tree species as terms.

SPECIES	MORTALITY
Eucalyptus miniata	2.6 + 0.7
Eucalyptus tetrodonta	3.2 + 0.9
Erythrophleum chlorostachys	5.2 + 1.9

DISCUSSION

Fire Behaviour

The environment of northern Australia is extremely fire-prone. This is because of the combination of a regular wet season which makes annual production of fine fuels possible, a long dry season which makes fuels flammable for all but a few months of the year, and the existence of potential ignition sources from either human or non-human sources over the whole of period when fuels are combustible. Thus, fire is more frequent in the eucalypt forests and woodlands of northern Australia (every 1-2 years) than in those of southern Australia (every 5-20 years: Cheney 1976; Stocker and Mott 1981; Walker 1981).

Measured fire intensity ranged from <500 to 18 000 kW m^{-1}; the latter was the most intense fire measured in northern Australia. Fire intensity at Kapalga was low, however, relative to potential peak intensities of *ca* 100 000 kW m^{-1} which may occur during wildfire in the forests of south-eastern Australia (Gill and Knight 1991). In the present study fire intensity varied with season of ignition. Over the years 1990-1992, the average intensity of fires in the early dry season (June) was less than that of those in the late dry season (September), which is consistent with the seasonal pattern of both Forest and Grassland Fire Danger Indices as determined by Gill *et al.* (1987, 1990). This is due to a number of factors. By the late dry season there is more fine fuel, primarily as a consequence of increased leaf-fall in the trees, both deciduous and evergreen. Afternoon relative humidity decreases progressively throughout the dry season, thus increasing the potential rates of spread (Gill *et al.* 1987; Gill and Knight 1990). However, despite the clear association between season of burn and fire intensity, in the present study there was substantial variation in fire intensity, both temporally and spatially (see also Bowman 1988) indicating that season of burn is not a precise predictor of fire intensity.

Effects of Fire on Savanna Trees

There are some aspects of this study which must be taken into account when assessing the effects of fire on savanna trees. In this study, as in that of Lonsdale and Braithwaite (1991), the effects of fire on the savanna trees have been determined by a single-time study of forest condition after fire, but without pre-fire measures of forest condition. Further, this study is in effect one of assessing the effects of fire following a set of relatively intense fires in the first year of the experimental burning - 1990 - and a set of generally low intensity fires in 1991. The fires of 1991 were of much lower intensity (5-15 times less intense) than those of 1990 in all but two cases. In these two cases, the fires of 1991 were about the same as those of 1990. Independent measurements of tree mortality on other reference plots following the 1991 fires indicate that tree mortality was minimal (R.J. Williams, unpublished data). Hence it is highly likely that the main patterns of stand condition measured on the experimental compartments at the end of the 1991-1992 growing season reflect the effects of the fires of 1990, rather than the combined effects of the fires of 1990 and 1991.

The effects of fire on patterns of tree mortality depended on interactions between fire intensity, tree size and tree species. Of the fire variables (fire intensity and season of burn), intensity was a more powerful predictor of overall tree mortality than was season, which explained less than 5 per cent of the variability in tree mortality. This does not imply, however, that season is unimportant, and the interaction between season and intensity is likely to be ecologically significant. For example, in most woody species the majority of flowering and leaf flush occurs between September and November (Brock 1988; Wilson *et al.*

1995), hence fires in the late dry season are likely to impact on these processes.

Tree mortality was also affected by tree size and species. Mortality increased with decreasing tree size, and was lower in the dominant eucalypts - *E. miniata* and *E. tetrodonta* - than in canopy subdominants such as *Erythrophleum chlorostachys*. Thus, fire has considerable potential to affect the species composition of savannas, given such differential susceptibility to fire of the major trees, both by species and by size. The reasons for this are at present unclear, but the relationship between bark thickness and species susceptibility are currently being investigated.

Management Implications

The major management implications derived from this study are that, first, season of fire is not necessarily a precise predictor of fire intensity; in some years, relatively intense fires can occur in the early dry season. The conditions determining these fires, and their degree of predictability, require further research. Second, single intense fires may have dramatic and long-lasting effects on forest structure and composition. There is some evidence that following the most intense fires of 1990, regeneration in tree saplings (less than 3 m tall) was stimulated on all burnt compartments. On the other hand, rates of recruitment from this below-3 m fraction of the tree population to the above-3 m fraction has been insufficient over the 1991-1993 period to replace losses due to the fires of 1990 (R.J. Williams; unpublished data). The old-growth stems of adult trees, and the actively growing sapling stages are both important components of long-term pools and fluxes of both nutrients and carbon. Increased loss of these components of the forest due to high intensity fire may lead to depletion of nutrient supplies (especially nitrogen) and a decreased store of carbon (see G. Cook *et al.*; this volume).

It is virtually impossible to prevent fire from occurring in the savannas of northern Australia. Therefore, the occurrence and impacts of different fire regimes on forest regeneration, given the variation in both timing and intensity of fire, requires further study, to understand the impacts of both fire regimes, and individual fires. In particular, there is need for long term monitoring of both post-fire seedling establishment and tree mortality; such studies are in progress at Kapalga.

In fire-impact studies, especially where season of burn is a factor in the design, measurements of fire behaviour are critical to interpretation of biological response. Direct measures which reflect the amount and rate of heat energy released in the vicinity of plants are discussed elsewhere in this volume by Burrows and Tolhurst. However, fire intensity - an integrated measure of energy release - is a convenient measure of fire behaviour, suitable for large-scale fire experiments such as the Kapalga experiment. Equipment such as that described by Moore and Gill (this volume), and the empirical relationships between intensity and post-fire attributes such as scorch height, are useful aids in this task.

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Quantifying bushfires for ecology using two electronic devices and biological indicators

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ABSTRACT

The main fire input to models concerned with the ecological impact of fires on biota or soils is temperature, either as a critical instantaneous value or as a time sequence. An instrumented system for measuring temperature-time curves in fires is described. Based on shielded mineral-insulated thermocouples and a data logger, it has proven to be effective, rugged, reliable and portable in a wide range of vegetation types. This system is relatively expensive. To obtain more extensive coverage of fires, less expensively, and to also measure rates of spread of fires, a 'temperature-residence-time meter' (TRTM) has been developed. This meter records the times that temperatures persist over a chosen value. By analysing 106 temperature-time curves from near ground level during fires in woodlands and forests at Kakadu National Park, we have found that the durations of chosen high temperatures were statistically intercorrelated ($r>0.82$) but that the times to reach peak temperatures from 60°C (a measure of flame residence time) were poorly correlated with these durations ($r>0.40$). Both electronic devices have widespread application for studies of fire ecology. Selected botanical attributes may be used as post-hoc indicators of fire properties.

INTRODUCTION

While ecological effects of fires are the result of the interactions between ecosystem properties and fire regimes, immediate impacts of fires can be related to the severity of fires as measured by, for example, fire intensity. While fire intensity - a measure of rate of heat release - is a most useful measure of fires for ecological purposes, it is a correlative measure rather than an explanatory one. Explanatory models of immediate fire effects invariably use fire-induced temperatures as inputs. Why this is important is indicated by various temperature thresholds *approximating* critical levels for various processes to achieve particular significance:

- 60°C, denaturation of proteins, hydrated-cell death;
- 100°C, boiling point of water, temperature of thermal arrest, desiccation of tissues;
- 300°C, ignition temperature, decomposition of plant materials, charring of tissues;
- 500°C mineralization of organic matter.

In all cases, these thresholds vary somewhat and are affected by duration of exposure. In leaf-scorch models, an instantaneous value of temperature for leaf death has often been used (e.g. Van Wagner 1973) or assumed, while a model of bark death in *Eucalyptus* (Gill *et al.* 1986) used a 60°C temperature of cell death and the time the external bark temperature persisted above 100°C as inputs. Mercer *et al.* (1993) used a lethal temperature of 70°C for seeds (after Bradstock *et al.* 1993) and various forms of external time-temperature curves as inputs to a model for predicting the impact of fires on survival of seeds in woody fruits. In a soil model, Aston and Gill (1976) used a temperature-time curve at the surface as an input.

Measuring temperature-time curves at various positions in the fire or plume is important for the provision of suitable inputs to ecological models. Here we describe a means of measuring the temperature-time profile during fires. As well, we describe an instrument which measures periods of time its sensor experiences above-threshold temperatures while recording the times of arrival and departure of the fire. The times of arrival of fires at a number of sensors can be used to measure rates of spread of fires. Rate of spread is a critical component of intensity measurement. The profiling system allows intensive sets of measurements to be taken while the cheaper instrument measuring durations of elevated temperatures can be used to measure variation across landscapes. Interpreting fire conditions can be aided also by measuring biological attributes which are discussed below.

TEMPERATURE-TIME PROFILES

The temperature-time profiling system first developed by the authors during 1988 is an effective, rugged, portable, minimal cost, reliable system which is easily and quickly put in place in the field. It uses thermocouple sensors displayed in vertical and horizontal arrays connected to a buried data logger. The thermocouples chosen for routine use were stainless-steel sheathed, mineral-insulated, chromel-alumel thermocouples (Type K, from 'Pyrosales Australia') 1.5 mm outside diameter encasing insulated wires of 0.25 mm diameter. The system has been used successfully in fires with intensities up to nearly 20 000 kW m^{-1}. Using this system, temperatures have been measured during fires in heathlands, mallee, temperate forests and tropical woodlands.

Thermocouples, unlike paints or crayons, give the time course of temperatures. Gill and Knight (1991) pointed out that temperature measurement by thermocouple, strictly speaking, gives a 'thermocouple temperature' rather than a 'fire temperature' because there are likely to be non-equilibrium exchanges of heat taking place between fire and thermocouple. Using the same equipment each time allows legitimate comparisons to be made. However, caution in interpretation is needed when comparing temperatures from different sets of equipment.

A typical temperature-time curve produced from the instrument is depicted in Figure 1: it shows the usual rapid rise in temperature as the fire reaches the thermocouple and the slower decline in temperature associated with the passing of the flames. The example is from a height of about 10 cm but the usual deployment of the 9 thermocouples in the vertical direction is from near ground level to a height of 9 m.

The temperature-time profiling system has now been used for five years in Australian vegetation types. The system used has similarities to those used overseas by Bidwell and Engle (1990), and Jacoby *et al.* (1992). However, these instruments recorded temperatures up to only 3 m.

The temperature-time profiling system provides high quality data for a single profile in the fire but, as the system requires a data logger and computer, it is relatively expensive. Because fire attributes often vary widely from point to point within a fire, additional measurements that can be taken over a wider area and less expensively, would be useful. The temperature-residence-time meter (TRTM), described below, is a novel instrument that partly achieves this objective.

THE TEMPERATURE-RESIDENCE-TIME METER

'Residence time' is one time measure in the time-dependent weight-loss curve for fuel, a curve which reflects the heat generation rate of the fire. 'Residence time' has been defined as the period of flaming combustion (Cheney 1981). However, as mentioned above, ecological studies may be concerned with times above certain temperatures, if not the details of the temperature-time curve, and these temperatures may be influenced by charring combustion and residual heat in the environment. Rothermel and Deeming (1980) suggested that the flame-residence time is equal to the time from initial temperature rise to the time of 'definite drop' following attainment of peak temperature whereas the temperature-residence times are longer (Fig.1). The TRTM's measure the time registered by the thermocouples above a certain defined temperature as well as rates of spread of fires.

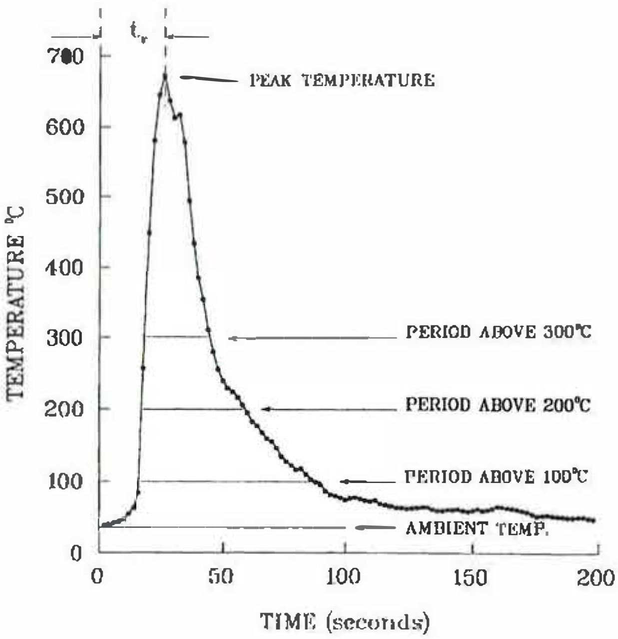

Figure 1. A typical temperature-time curve for surface fires. 'Time' is the elapsed time from the first rise above ambient. t_r is the flame residence time (Rothermel and Deeming 1980).

The TRTM, is a small instrument that is buried in soil with a thermocouple sensor exposed above ground in the path of a fire; it records the times of fire arrival at, and departure from, the sensor. The prototype was developed in 1988. By having three or more meters laid out in triangles (Simard *et al.* 1984), rates of spread can be calculated from the arrival times of the fire at the apices of the triangles.

The fire meter is comprised of a small plastic box (12 cm x 6.5 cm x 4 cm.) incorporating a digital stopwatch and electronic circuits, a connector for a detachable thermocouple, and external electronic contacts (for batch battery charging of meters in a 'holding box' and for synchronizing the starting and resetting of all meters). The detachable thermocouple consists of 1 m of thermocouple wire on a 60 cm lead of thermocouple extension wire.

The meter is triggered (on or off) by a thermocouple-generated voltage equivalent to an approximate thermocouple temperature of 200°C. This thermocouple temperature was chosen as being high enough to avoid triggering the meter in air and appropriate to indicate the presence of flames when placed 10 cm above the ground. The threshold level is 'approximate' because '200°C' really represents the difference in temperature between the exposed thermocouple and the cold junction buried in the soil; the error due to variation in soil, and cold-junction, temperature is relatively small. The threshold temperature can be adjusted to the user's specifications.

The thermocouples chosen for routine use were the same as those used for the temperature-time-profiling system. They were effective, rugged and reliable. Fused bare-wire thermocouples of the same 0.25 mm diameter continually broke down, short circuited or turned the instruments off prematurely (due to their sensitivity to cooler-air pulses within the fire as the flames died down). A finer-drawn version of the sheathed thermocouple with a 1 mm outer diameter was found to be satisfactory in terms of ruggedness and reliability on most occasions but there were more breakdowns with this than with the larger-diameter version. Because of the detachable nature of the thermocouples, users can choose the thermocouple most appropriate to their particular needs.

TRTM is a reliable, relatively inexpensive, portable, reusable instrument suitable for grass and litter fires of intensities up to at least 18 000 kW m^{-1}.

We have analysed 106 temperature-time curves from thermocouples placed 5 to 10 cm above ground during fires in woodlands and forests in Kakadu National Park. The durations of temperatures above 60, 100, 200, 300 and 400°C were all intercorrelated with statistically significant correlation co-efficients between 0.82 and 0.96. The period from the time the thermocouple reached 60°C to the time it reached peak temperature - a measure of flame-residence time - was highly significantly correlated with the other times recorded above but with the relatively low correlation coefficients between 0.40 and 0.54. Thus, there is a trend indicated between 'flame residence time' and 'temperature residence time' but prediction of one from the other on the basis of these measurements is inappropriate.

For rate of spread measurement, the TRTM, is a development of the instrument reported by Blank and Simard (1983). The latter workers used a piece of solder to sense the fire (by melting). TRTM may be reused without sensor replacement, is easier to read, and has greater reliability and sensitivity than Blank and Simard's instrument; it is more expensive but still of a reasonable cost.

FIRE INTENSITY AND TEMPERATURE

Fire intensity has become a standard variable to measure in studies of fire ecology so it is of interest to examine the relationship between intensity and temperature. Intensity is a measure of the rate of heat release per length of fire-line (fire-line intensity, I_B, Byram 1959) or burning area (reaction intensity, I_R, Rothermel 1972).

$$I_R=I_B.d$$

where d is the flame depth.

$$I_B=H.w.r$$

where I_B is the intensity in kW m^{-1}, H is the heat of combustion in kj kg^{-1}, w is the fuel loading in kg m^{-2} and r is the rate of spread in m sec^{-1}.

Current theory links intensity, I, and peak temperature of the temperature-time curve, T_m, at different heights, z, but only in the plume well above flames (Yih 1952; Van Wagner 1973):

$$T_m - T_a = k(I^{0.67}/z)$$

where T_a is the ambient temperature and k is a constant. Recent work by Weber *et al.* (1993) examines the problem of extending equations to the prediction of temperatures in the flaming zone. The relationships in the equation above appear to have limited value for ecological studies because of their applicability to the plume of the fire only and because temperature duration is not predicted. However, in the above-mentioned studies by the authors at Kakadu National Park, correlations between peak temperatures and durations of temperature above 60, 100, 200, 300 and 400°C in the 106 curves analysed were statistically significant with correlation coefficients between 0.64 and 0.82.

BIOLOGICAL INDICATORS OF TEMPERATURES

Post-hoc indicators of fire severities for ecological studies are useful because fires may occur unexpectedly in study areas and intensity limits to fire control (Luke and McArthur 1978) may limit experimentation. The 'indicators' can be the items of interest to the ecologist but to put the situation into a management context may require a knowledge of the links between fire properties and the 'indicators' themselves. Common indicators are height of leaf scorch (Fig.2) while less commonly considered are thickness-profiles of bark lost from smooth-barked trees (Fig.3; Gill 1981) and heights of leaf char (Fig.4). Diameters of live stems consumed by fire at different heights provides another indicator (Fig.5). When considering biological indicators it is important to remember that, apart from the fire itself, immediate prefire tissue temperatures and moisture contents and species effects may be important in determining the observed results.

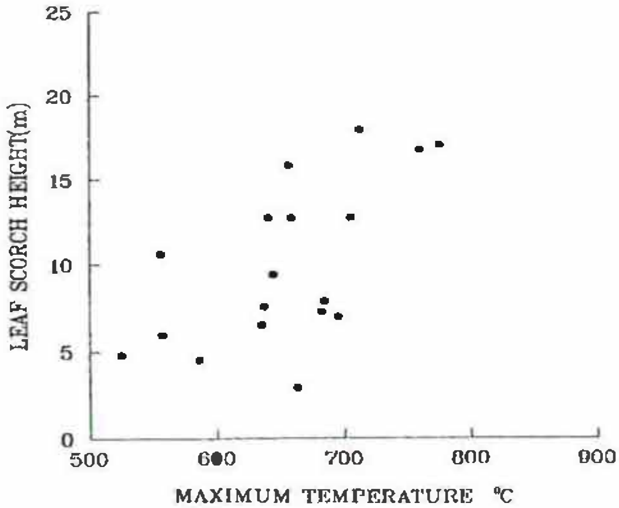

Figure 2. Maximum height of leaf scorch as a function of maximum temperature reached in temperature-time profiles during fires at Kakadu National Park in 1990-1991.

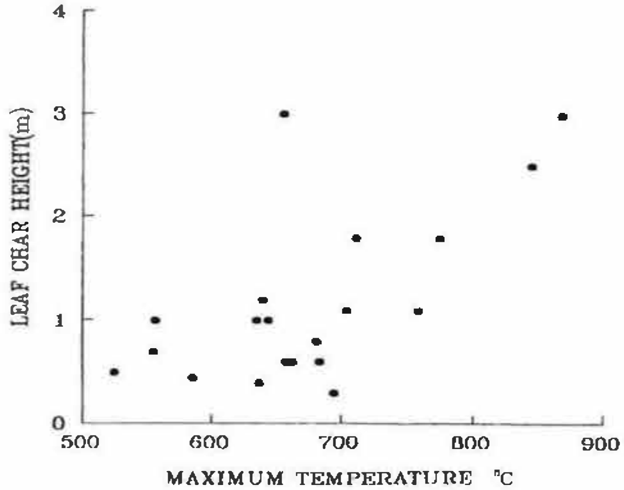

Figure 4. Maximum heights of leaf charring as a function of maximum temperature reached in temperature-time profiles during fires at Kakadu National Park in 1990-1991.

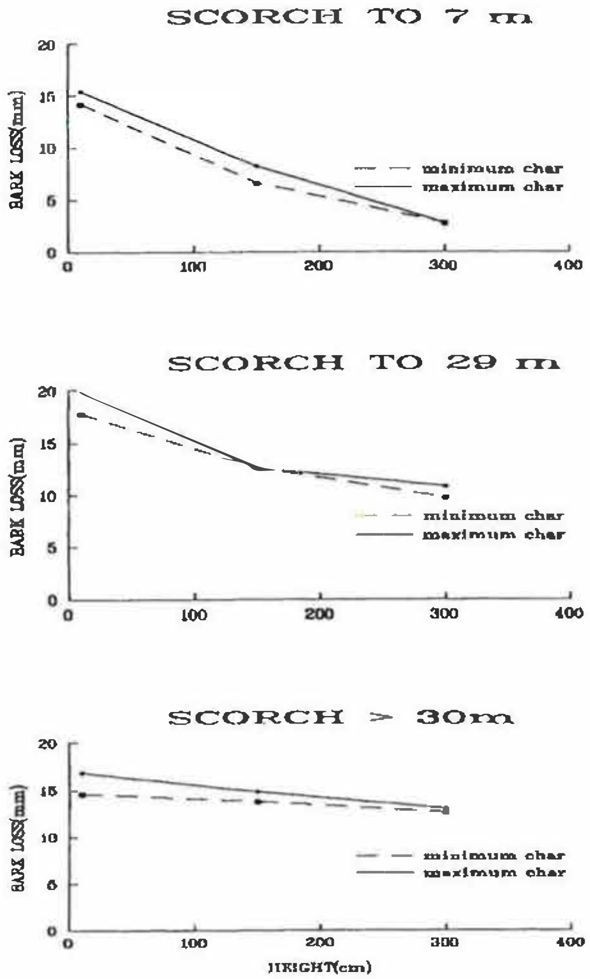

Figure 3. Thicknesses of decorticated bark as a function of sample height on mature smooth-barked trees following fires. The examples, all from south-eastern Australia, were chosen to show bark losses in fires producing a range of scorch heights.

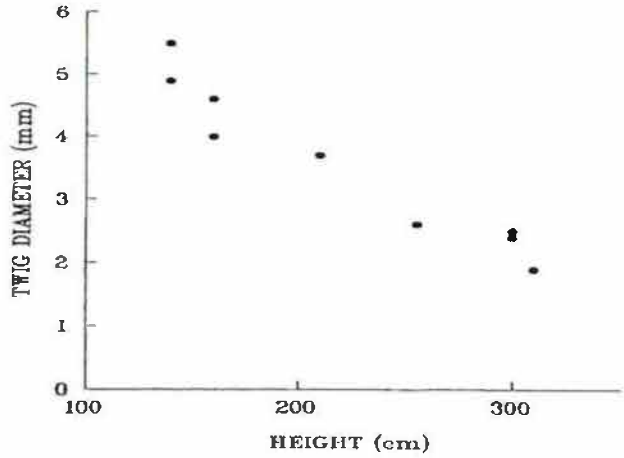

Figure 5. Diameters of live stems, twigs or branches of mallee eucalypts consumed by fire at various heights during fires at Yathong Nature Reserve, New South Wales, December 1991.

CONCLUSIONS

Two useful instruments with widespread application have been described and thoroughly tested. The temperature-time-profiling system measures profiles of temperatures for the duration of the fire. The temperature-residence-time meter measures the times of arrival and departure of a fire at a thermocouple sensor and records the duration of exposure of the sensor above a threshold value. From triangulated TRTM measurements, the rate of spread of a fire can be measured. The authors have found the instruments to be rugged, reliable, portable and effective. In the absence of such instruments, biological indicators may be used but their study is in it's infancy.

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Using technology to facilitate community preparedness (abstract only)

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ABSTRACT

Fire authorities constantly strive to improve their capacity to control bushfires, using the results of inquiries and research. Significant gains have been achieved in strike power and organisation have been achieved following major bushfires. Despite this, however, significant life and property loss are still likely in major bushfires in the urban interface.

This paper considers the public warning and importance of providing knowledge of the bushfire phenomena and real time information on the immediate fire threat. It reviews a new Country Fire Austhority (CFA) project, Community Fireguard, which targets people in high fire hazard areas with a view to getting them to accept responsibility for their own bushfire safety. The CFA's technological thrust using sophisticated computer technology is also considered in the context of being better able to provide real time fire spread information to the threatened public.

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Analysis of rural ignition patterns on Canberra's urban/rural interface

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ABSTRACT

An approach to using records of man-made ignition patterns, as part of the overall system for rural fire hazard management in the ACT, is presented. The area being managed has been divided into blocks exhibiting a relatively uniform land-use pattern. A predictive model relates expected number of ignitions per square kilometre per annum to distance from suburbs and principal land-use.

The relationship between ignition rates, land-use types and distance from nearest suburbs shows that Canberra Nature Park, industrial lands, commercial sites and pine plantations all have relatively frequent ignitions (expressed as number of fires per square kilometre per annum) close to suburbs. Ignition rates in the Nature Park and commercial sites decline rapidly as distance from nearest suburbs increases. Rural lands have constant, low ignition rates.

Some implications of these patterns are discussed. An urban/rural interface transition zone can be defined using the model. The model allows improved liaison with planning authorities and land managers. It also offers a quantitative method for setting standards of fire response.

INTRODUCTION

The urban/rural interface of Canberra is a major problem for fire managers and land managers. The basic planning philosophy of Canberra as a 'bush capital', with abundant open space (Seddon 1977), has led to discrete town centres being developed, separated by hills and ridges that are left undeveloped. The growth of the city is by means of the building of largely contiguous new suburbs, each of which replaces traditional rural land-uses. There is thus a very convoluted and long interface[1]. A recent trend towards urban consolidation may do little to counter this. The interface is almost entirely classic [clearly demarcated transition], with some occluded interface [rural intruding into urban] and very little mixed interface [urban outliers in rural areas] (using the terms of Laughlin and Page 1987).

As part of an assessment of rural fire hazard in the ACT (McRae 1991), the following process was carried out:

- Define the problem to be addressed. Here rural fire hazard was defined as being:

Hazard: *A measure of how frequently fires occur, how quickly they spread and how close they are to life and property.*

This may be restated from an operational perspective as being a need to provide quick detection, quick response and quick suppression.

- Devise a process that allows a quantitative or semi-quantitative assessment of hazard. This was done through a flow chart that led from raw data through to modelled indices (Fig. 1).
- Devise ways of using the assessment to guide management and planning for land-use and fire. If the hazard were assessed as being high, then working backwards through the flow chart provides an indication of the most effective ways of mitigating the hazard - and also indicates which factors are not able to be managed (such as being prone to natural - i.e. lightning - ignitions, see Figure 2). Further, the modelling software indicates which of the component indices contributed most to the hazard value. These guides permit cost-effective hazard mitigation.

Achieving the relatively straight-forward process outlined above proved quite difficult in a number of respects. The prediction of areas prone to lightning ignition required a new procedure to be developed

[1] With a population of over 290 000 in 1993, Canberra had an interface estimated to be 1400 km in length.

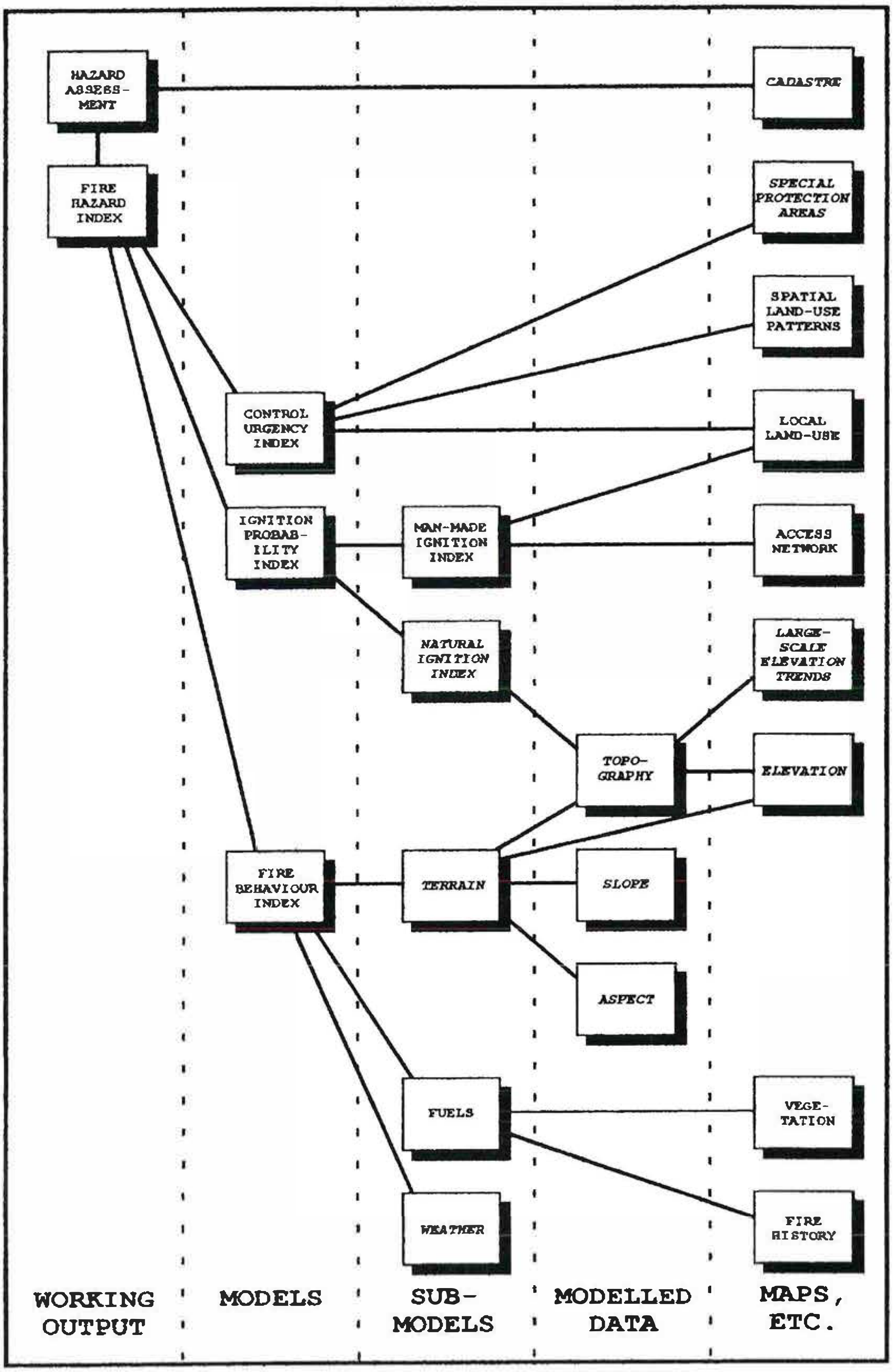

Figure 1. Flowchart for ACT rural fire hazard assessment. Those boxes labelled in italics show factors that cannot be managed to mitigate hazard levels.

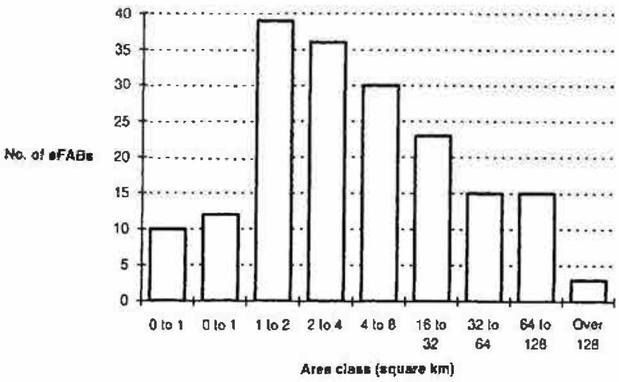

Figure 2. Area distributions for sub-Fire Accounting Blocks in the ACT.

(McRae 1992). There was little known about the spatial distribution of ignition patterns around Canberra's urban interface, or even their distribution among generic land-use types. The ACT Bush Fire Council's fire records archive proved an invaluable and almost complete source of information, allowing many of the problems to be solved.

The full-time emergency agencies in the ACT are now combined as the Emergency Management Group, with agency identities retained, but centralizing of all support roles. This group is revitalizing the Emergency Planning process, and has adopted an 'all-hazards' approach to finding the balance between:

- overall threats to the community;
- agencies' capabilities to respond to emergencies;
- the need for cost effectiveness.

METHODS

As part of the assessment of rural fire hazard in the ACT, and the on-going establishment of a quantitative foundation for the new Emergency Management Group, a number of steps have been taken towards a quantitative analysis of fire ignition patterns. This paper deals with one aspect of this work - the detailed relationships between ignition frequency and land-use and proximity to suburban development.

The ACT was divided into a series of blocks with relatively uniform land-use patterns, called Fire Accounting Blocks (**FABs**). Some of these were divided further, by progressing into a finer scale of resolution, into sub-**FABs** (**sFABs**). There were 182 of these. For each **sFAB**, the area in square kilometres was calculated, and the number of recorded fires over the last 11 years was noted from operational archives. The observed average number of fires per square kilometre per annum, the Man-made Ignition Index ($\mathbf{MII}_{Obs}$), was calculated as the fire tally divided by the area divided by 11, the number of years of observation. Changes in land-use during the time interval were taken into account. The frequency distribution of **sFABs** in area classes is shown in Figure 2.

The principal land-use (**PLU**) for each **sFAB** was determined from maps, air photos and office records. A distance (**DIST**), in kilometres, from the centre of each **sFAB** to the nearest edge of suburban development was calculated.

A line of best fit was obtained which, for each **PLU**, could predict ignition frequency ($\mathbf{MII}_{Pre}$) given **DIST**. The equation for the model is:

$$\mathrm{MII}_{Pre} = e^{a*} \quad e^{(b* \ \mathrm{DIST})}$$

which was reduced to a linear form that allowed easier fitting of the constants:

$$\log_e(\mathrm{MII}_{Pre}) = a + b* \quad \mathrm{DIST}$$

The values of a and b were determined for each principal land-use. Unfortunately, low sample sizes prevented formal regression analysis, lines instead being fitted to the data by eye.

RESULTS

The values of the constants are shown in Table 1 and the curves generated by the model for each land-use are graphed in Figure 3. The general nature of the equation is that with increasing distance from suburbs the ignition frequency starts from an initial value and declines towards zero. The initial value varies widely, as does the rate of decline. In one case the rate of decline is zero, i.e. the value is constant.

In general, small sample sizes ruled out a quantitative assessment of the goodness of fit for the model. However, the general trends are clear enough for the model to be useful. Further work is required to improve the statistical rigour of the model.

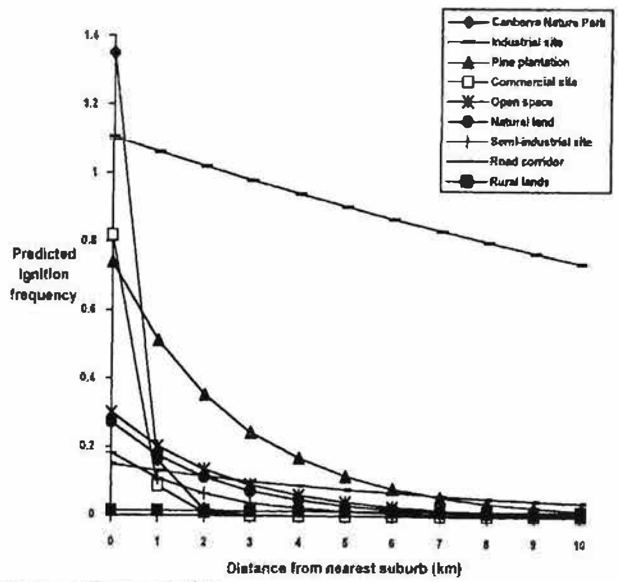

Figure 3. The relationship between predicted ignition frequency and distance from nearest suburb (ranging up to 10 km) for each principal land-use.

TABLE 1

The areas of each principal land-use type, together with the values of the two constants used in the predictive model for ignition frequency. Also shown for the land-uses are categories (zero, low, moderate and high) describing the relative initial value of ignition frequency and its relative rate of decline with distance from nearest suburb.

PRINCIPAL LAND-USE	AREA (km²)	VALUE FOR CONSTANT A	VALUE FOR CONSTANT B	INITIAL VALUE OF IGNITION FREQUENCY	RATE OF DECLINE OF IGNITION FREQUENCY
Canberra Nature Park[a]	56	0.3	-2.1	High	High
Industrial site	16	0.1	-0.04	High	Low
Pine plantation	235	-0.3	-0.4	Moderate	Moderate
Commercial site	15	-0.2	-2.2	Moderate	High
Open space	30	-1.2	-0.4	Moderate	Moderate
Natural land	1090	-1.3	-0.4	Moderate	Moderate
Semi-industrial site	10	-1.7	-0.5	Low	Moderate
Road corridor	93	-1.9	-0.1	Low	Low
Rural lands	526	-4.3	0	Low	Zero
Urban	150	n.a.	n.a.	n.a.	n.a.

[a]C.N.P. is a series of discrete natural or semi-natural land parcels in and around the city, managed for recreation and conservation.

APPLICATIONS - RESPONDING TO PATTERNS IN IGNITION FREQUENCIES

The predicted ignition frequency model assumes that all of the spatial pattern in ignition frequencies can be related to distance from closest suburb and principal land-use. To interpret this, the suburbs may be viewed as a source of ignition in the form of people going into nearby open-space and related areas for various purposes, but in the process causing fires, either maliciously, carelessly or accidentally. The destination can be either 'whatever is on the other side of the fence', or some specific attraction, such as a favoured picnic spot or a site favoured because arson may be carried out with little risk of detection. However, the magnitude of all of these effects declines with distance from suburbs.

Among the issues arising from this that are now being addressed in the ACT are:

(1) Definition of an interface transition zone.
(2) Liaison with planning authorities.
(3) Liaison with land managers.
(4) Standards of fire response.
(5) Decision support for dispatch.

Definition of an Interface Transition Zone

The predicted ignition frequency patterns can be used to define an urban/rural interface transition zone. Rather than addressing the interface itself, this approach considers the transition zone on the rural side of it where rural fires are most frequent.

The zone can be defined as including all points where predicted ignition frequency exceeds a specified value. (Lands closer in to the suburbs with different principal land-use may be outside the zone.) Two approaches to the cut-off value examined are:

- Predicted ignition frequency is 50 per cent of what it would have been if that land were adjacent to suburbia.
- A constant value for predicted ignition frequency (such as 1.0, 0.5 or 0.25 fires per square kilometre per annum).

Table 2 shows the results of these.

From the perspective of fire management in the ACT, no formal definition has yet been adopted, but that based on a predicted ignition frequency of 0.25 fires per square kilometre per annum appears the most valuable as a starting point as it quantifies existing working arrangements. The zone resulting from this is identified in the map in Figure 4.

TABLE 2

Definitions of the interface transition zone, based on distance from suburbs to achieve specified ignition frequencies. All distances are given in kilometres.

PRINCIPAL LAND-USE	DISTANCE TO 50 PER CENT OF MAXIMUM FREQUENCY	DISTANCE TO 1.0 FIRES PER km^2 PER ANNUM	DISTANCE TO 0.5 FIRES PER km^2 PER ANNUM	DISTANCE TO 0.25 FIRES PER km^2 PER ANNUM
Canberra Nature Park	0.33	0.14	0.47	0.80
Industrial site	17.31	2.5	19.83	37.16
Pine plantation	1.88		1.06	2.94
Commercial site	0.90		0.22	0.53
Open space	1.74			0.47
Natural land	1.58			0.20
Semi-industrial site	1.32			
Road corridor	5.34			
Rural lands	n.a.			
Urban	n.a.			

Figure 4. Map of the ACT urban/rural interface transition zone (shaded black). The outlined areas are suburban development.

Land-use Planning Guidance

The model can be used to guide cost-effective land-use planning. Planning authorities are constantly attempting to balance a broad-spectrum of competing demands in urban or rural environments. One of these, protection from wildfire, has rarely been satisfactorily quantified, and has thus often been given a lower priority than factors such as visual and recreational amenity or traffic flow. Changes in public perceptions and the threat of massive economic impact following large wildfires are forcing a change in both the effort expended in addressing the threat and in prioritizing protection measures.

Obvious strategies suggested by this include:

- Conducting an assessment of the overall impact on the local fire problem arising from a development proposal, allowing cost effective fire protection and forecasting of impacts on the workloads of fire agencies.
- Liaising with planners to ensure that lands adjoining new suburbs are not those with the highest predicted ignition frequency. For example, general open space is better from this perspective than a pine forest.
- A 'buffer' of less ignition prone lands between suburbs and other land-uses could prove effective. A buffer width of 200 m would significantly alter ignition frequencies. Buffers would also improve fire protection for the interface, and, if properly designed, could reduce maintenance costs - for example, by reducing the need for fire fuel management.

Land Management Agency Liaison

The on-going liaison between Rural Fire Service and Government land management agencies and rural elements of the private sector benefits through use of the model to allow full recognition of the potential impacts of the ignition patterns. The need for plans of management to be prepared or the threat of economic impact drives the timetable for this process.

A number of measures are currently in place in the ACT, including:

- Liaising with land management agencies to ensure that their management goals are linked into ignition patterns. As an example, of two similar sites for a rare plant species, one closer to suburbia will have less chance of long-term survival since it has a higher ignition frequency. Both sites may, however, be given an equal chance of survival by expending more fire suppression or fire protection effort to the area closer to suburbia. However, this requires an on-going commitment and may have a large impact upon a limited budget.
- Where the land management agencies provide fire suppression resources, the deployment of these is optimized with respect to proximity to current ignition patterns, minimizing travel times, and management of the total costs of having resources committed to fire duties.
- We are seeking a better understanding of the reasons when the observed ignition frequency differs greatly from the predicted ignition frequency. Low observed values may be due to some aspect of local land management. For instance, some lands in Canberra are managed as part of the seat of Federal Government and are not used by the public in the same way as other lands. Also, the results of changes in land management may take a number of years to show up clearly in the observations. Finally, localized increases in ignition frequencies may be due to the actions of arsonists, and we are working closely with the Australian Federal Police on utilizing these observations for law enforcement.

These measures may suggest locally applicable ways of preventing fire, but they must also be compatible with any current plans of management. For example, closing an access road into an area of high recreational demand would reduce fire frequency in the area, but this may be counter to public expectations expressed during the preparation of the plan of management.

Standard of response

The predicted ignition frequency pattern can be a cornerstone of cost-effective rural fire protection. The interface transition zone can be used to determine a specific standard for response, based on detection, actual response times and number of units dispatched. Each component of the standard can be optimized with respect to both the standard and cost.

Table 3 highlights some of the problems posed by the transition zone in comparison with other rural lands.

TABLE 3

Comparison of characteristics of the transition zone and other rural lands.

COMPONENT	TRANSITION ZONE	OTHER RURAL LANDS
Ignition frequency	High, rapidly decaying with distance from nearest suburb.	Low, approximately constant, natural ignition patterns prominent.
Detection	Good - '000' calls from public.	Poor. Need commitment to staffing fire tower network.
Access	Good. Many access points from suburbs, frequent vehicular tracks. Some hindrance from drains, etc.	Sparse rural road network, few access tracks. Need to travel cross-country.
Access strategy	Urban units radiate out from urban centres. Rural units deployed at key points on rural periphery, or converge from outer rural areas.	Local units nearby, other units have large distances to travel.

Identification of a separate transition zone permits separate standards of response to be applied:

Transition Zone: As the fire danger index goes up, required response times become shorter, reflecting the expected increase in potential rate-of-spread. Note that in the ACT there is a lower limit of perhaps 10 minutes which cannot be bettered without undue cost to the community.

Other: Response times for first and second units, and their destination with respect to incident, should reflect the value of the Control Urgency Index (**CUI**, McRae 1991), as indicated in Table 4. Control Urgency Index is a component of the ACT Rural Fire Hazard Assessment and reflects proximity to property or resources.

These times could be improved considerably as fire danger index increases and units are strategically deployed, but the degree of improvement that can realistically be achieved represents a balance between the cost of standby and perceived threat to the community.

TABLE 4

Goals for Standards of travel time outside the interface transition zone.

CONTROL URGENCY OBJECTIVE	INDEX	1ST UNIT	2ND UNIT
5	20 minutes	30 minutes	At fire
4	20 minutes	45 minutes	At fire
3	30 minutes	1 hour	At fire
2	45 minutes	2 hours	In area
1	1 hour	As needed	In area

Decision Support for Dispatch

Software has been developed, and is being refined, that uses all of the above elements, and the details of timing and location from the Rural Fire Service standby roster, to advise on the most appropriate units to dispatch to any location. This can be the key to ensuring that the standards of response are met as often as possible.

CONCLUSION

This analysis of historical data for the ACT has confirmed the conventional, subjective opinion that most fires occur close to the urban/rural interface. A clear pattern emerged relating land-use and distance from nearest suburb to the frequency of ignitions.

At a time when the ACT's urban and rural fire services are having to learn how to work together more closely this finding is of great benefit.

The Emergency Management Group has recently redefined the legal definition of the fire services' jurisdictions to reflect land-use planning policies in the Territory's key planning instruments, the Territory Plan, developed by the ACT Government's Territory Planning Authority, and the *National Capital Plan*, developed by the Federal Government's National Capital Planning Authority.

As jurisdiction is now based on land-use, and land-use is being used to quantify fire management on the urban/rural interface, we have been able to resolve what has been a major issue for interface fire protection.

There were insufficient data on ignition patterns to permit proper statistical analysis. Forthcoming fire seasons will see data being collected specifically for this purpose. There is a continuing dilemma in the ACT - the balance between resolution and precision. The area is so small that a compromise must be reached between dividing any data set up to resolve the interaction of determining factors and yet still having enough replicates to allow rigorous analysis. This applies in spatial and temporal analyses. A further problem with temporal analyses is that the growth of the city is so rapid that the extension of the duration of an historical analysis in order to gain extra precision must be cognisant of the fact that past trends may no longer be relevant.

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Towards an integrated model for designing for building survival in bushfires

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ABSTRACT

To date, there have generally been two approaches to mitigating building destruction. One approach has been to manage the vegetation (the 'landscape approach') whereas the other has been to select building materials and designs to minimize the effect of the bushfire attack (the 'building approach'). There is a need to combine these two approaches and this paper suggests that it can be done on the basis of the agents by which bushfires ignite, damage and destroy buildings, *viz.* burning debris, radiant heat, flame and wind.

A model to combine these approaches might be comprised of three modules:

(a) the first module is the potential bushfire attack, based upon the properties of the vegetation producing that attack;

(b) the second module is the modification of the attack by the environmental conditions and the landscape (this provides the opportunity to use the 'landscape approach'); and

(c) the third module is the reaction of building materials and design to the (modified) bushfire attack (this provides for the 'building approach').

This paper describes such an integrated model, briefly discusses the current state of knowledge with regard to the modules and the research that is needed to realize a quantitative model.

INTRODUCTION

A significant portion of the cost of bushfires in Australia is the destruction of buildings and the loss of life. For example, the 1983 Ash Wednesday fires in Victoria and South Australia caused the loss of 76 lives and damage estimated at $400 million which included some 2463 houses (Ramsay *et al.* 1986). Most of the buildings at risk were those situated at the interface between urban development and the countryside. Such development is increasing as cities and towns spread, lower-priced housing is needed, and the demand for recreational and permanent housing increases in areas where landscape and indigenous vegetation provide desirable environments. The nature of the countryside comprising the interface varies considerably and may include farms, orchards, forest plantations, national parks, state forests and undeveloped areas. However, the most desirable areas appear to be those with an abundance of indigenous vegetation, and consequent high bushfire potential.

These trends reinforce the need to give more attention to design for the survival of buildings in bushfires and for better information to facilitate such activities. This paper addresses the need for an integrated model, based on the underlying mechanisms which cause the ignition and destruction of buildings, which could be used to facilitate the design of buildings and their surrounding landscape.

MECHANISMS OF BUSHFIRE ATTACK

The mechanisms of bushfire attack, which have been established (Barrow 1945; Cole 1983; Ramsay *et al.* 1986; Wilson and Ferguson 1986) by examination of buildings (or their remains) which have been subjected to bushfires, are:

(a) ignition by burning debris (carried by wind);

(b) ignition and damage by radiant heat;

(c) ignition by flame contact; and

(d) damage caused by wind accompanying the fire.

Ignition by burning debris is generally believed to be the major mechanism and this may occur in a number of ways. The burning debris, along with other windborne combustible material such as leaves and twigs, can:

(a) pile up against combustible materials used at, or near, ground level as stumps, posts, subfloor enclosures, steps, door frames and window frames;

(b) accumulate on combustible materials used for decks, verandahs, pergolas and window sills;

(c) lodge in gaps and crannies in combustible materials used for exterior wall cladding, window frames and door frames; and

(d) gain entry to the interior of the building through broken windows (damaged by radiant heat, the force of the wind or flying debris), or through gaps in the exterior wall or roof cladding. Once inside the building, the burning debris may ignite framing, fittings and contents.

The small ignitions induced gradually get larger, involving other combustible parts of the building and its contents, until the building becomes totally involved in fire.

Heat radiated from a bushfire may assist ignition caused by burning debris. It may do this by preheating the building and its contents, or by breaking the glazing of windows, allowing burning debris to enter the building. In extreme cases, the heat may be sufficient to directly ignite the combustible exterior parts of the building and the interior furnishings situated near windows. Flame from the bushfire may also ignite combustible exterior parts of the building, particularly if vegetation located directly against the building catches alight.

Wind carries burning debris, driving it against or into a building, as well as large items such as branches and building materials which may break windows and damage roofing. The forces produced by the wind may also damage a building by breaking windows, removing portions of the wall and roof cladding or loosening them; this assists the entry of burning debris and leads to the ultimate destruction of the building.

CURRENT DESIGN APPROACHES

To date, there have generally been two approaches to mitigating building destruction. The first approach has been to manage the vegetation by means of fuel reduction and firebreaks. This approach has become more sophisticated with the use of measures such as windbreaks, suitable location of amenities and protective plantings, and is referred to in this paper as the 'landscape approach'. The second approach has been to select building materials and designs which were believed to minimize the risk of the bushfire damage. This 'building approach' tends to assume a certain level of bushfire hazard and 'designs against it'.

Present design processes tend to reinforce this dichotomy, with the building design and construction being separate and preceding any detailed consideration of the surroundings of the building and landscape design. When professional designers are employed, different people generally carry out the design of the building and the landscape, even if they are from the same practice, further enhancing the division of the design process. Ideally, for all the reasons discussed below, the two design processes should be carried out concurrently, enabling interaction between the two approaches. However, where this is not practical, it is preferable for at least the overall landscape design to be carried out *before* the building design. This would provide the opportunity to include desirable landscape measures, which may otherwise be precluded by the siting or design of the building.

THE NEED FOR A COMBINED APPROACH

Proponents of the landscape approach to designing (especially when confined to fuel reduction) have suggested that eliminating or reducing the bushfire attack at its source is the most effective and practical course of action, especially given the reluctance of builders and owners to incorporate recommended bushfire measures into new and existing buildings. However, to ensure building survival solely by this means would often necessitate a degree of fuel removal, which is unacceptable to many people, and remove much of the attractiveness of 'living in the bush'. Where allotments are small, such fuel reduction may require the cooperation of the owners of neighbouring allotments. In some cases the 'neighbouring allotments' may be extensive public lands such as forests and conservation areas where substantial landscape measures are neither possible nor desirable. Furthermore, burning debris may travel considerable distances and removing the vegetation in the vicinity of the building would not eliminate this hazard. Taken to its extreme, this is an argument for either the removal of most indigenous vegetation or its replacement by exotic vegetation. In addition, there is the problem of the long-term maintenance of such fuel reduction measures.

Proponents of the building approach take the view that useful modification of the landscape (especially vegetation) is not feasible either in the practical sense or politically, that the building itself should be built in such a way as to prevent its ignition and destruction and that people should be encouraged to stay in their buildings during a bushfire to ensure this. This approach becomes more and more difficult to accomplish as the fire hazard of the landscape increases and the cost of further building measures increases accordingly. The 'concrete bunker' is the ultimate outcome of this approach. In addition, there is no guarantee that people will be:

(a) with the building at the crucial time;

(b) be able to extinguish multiple ignitions before they grow to the stage that they are out of control; or

(c) aware of all the ignitions which have occurred.

Without a combined approach, landscape and building measures tend to be implemented independently, leading to either a reliance on one approach to the exclusion of the other or the use of both approaches regardless of the benefits of the other (a 'belt and braces' approach). In both cases, other requirements for the development such as retention of the vegetation amenity, passive solar design and large areas of glass for views may be sacrificed in the pursuit of building survival. This points to one of the strengths of a combined approach: the flexibility to accommodate other requirements by judicious choice of both landscape and building measures. This flexibility leads to a range of design solutions most appropriate to the development, and to possible consequent cost savings. For example, there may be a situation where there is a desire to have large windows facing a particular direction for aesthetic reasons and sunshine penetration into the building. The windows that satisfy these objectives could be a serious weakness in a bushfire because they could be cracked by radiant heat or broken by windborne objects, thus allowing burning debris to enter. Even if unbroken, they could admit radiant heat into the interior making combustibles more prone to ignition by embers.

There are many possible options to solve this conflict and protect the building:

(a) use special glass which is resistant to breaking;

(b) reduce the size of the panes of glass;

(c) use metal insect screens externally to catch embers and reduce radiant heat;

(d) use metal shutters (sliding, vertical, rolling or hinged);

(e) avoid locating any external combustible structures close to these windows;

(f) protect these windows externally with suitable planting located to form a barrier or deflector for wind, windborne debris and radiation;

(g) manage the vegetation for a given distance outside these windows to reduce ember, radiation and flame attack.

It will be seen that the first five options listed are *building* options and the last two are *landscape* options. Some of these options can be used in combination thus adding to the flexibility of a combined approach.

The combination of the landscape and building approaches model is most practically achieved by a consideration of the individual measures that might be applied for each building element. In the present state of knowledge, the matching of a finished landscape design with a complete building design is not yet possible as there is no quantitative measure of either with regard to bushfire hazard or bushfire performance respectively.

AN INTEGRATED MODEL

For the effective combination of the landscape and building approaches, it is desirable to have a model of the urban interface under bushfire conditions. Such a model, coupled with a bushfire spread model, could be used not only for the design of building and its surrounding landscape, but also for fire prevention and firefighting activities to minimize much of the building loss.

A Structure Ignition Assessment Model (SIAM) has been proposed (Cohen *et al.* 1991) which comprises three major modules for 'fire behaviour', 'heat transfer' and 'building ignition' with a subsidiary 'fire brand' module. This model appears to concentrate on ignition of combustible elements of the building exterior and does not include ready provision for the integration of landscape measures and design. In addition, 'heat transfer' by way of radiation and flame contact is a major module of the model rather than the 'fire brand' module, despite the assumption (Tran *et al.* 1992) that 'ignition by burning debris plays a very important role in the ignition of structures'.

We believe that a more appropriate model, based on the elements of bushfire attack (burning debris, radiation, flame and wind) might have three modules which:

(a) quantify the potential attack according to the vegetation types and quantities;

(b) modify the attack according to the environmental conditions and landscape features; and

(c) determine the response of the building according to its materials and design.

Bushfire Attack Module

Although there is an extensive body of literature (Gill *et al.* 1991) on bushfires and their effects, very little effort has been devoted to characterizing and quantifying the attack that the burning of various types of vegetation might generate in the vicinity of buildings. The literature was recently reviewed (Rudolph 1993a) and a qualitative approach developed for assessing the effects of individual species. Fourteen attributes were identified and each was considered separately for its effect on each of the four elements of bushfire attack (embers, radiation, flame and wind) (see Table 1). The Table shows, for example, that when considering the attributes of bark texture, increasing looseness (as opposed to tightness) tends towards an increase in ember attack. Although there are a large number of 'unknowns' in the Table, this work provides useful information for assessing bushfire attack and a systematic basis for accumulation of data for the future.

Much more data is required before a quantitative approach can be taken and viable models for bushfire attack developed. In the meantime, we must rely on experience and expert judgment to determine the likely modes and magnitude of attack that a given vegetation loading will produce.

TABLE 1.

Effects of attributes of plants on four modes of bushfire attack.

Effect of attributes on mode of attack
I = increase, D = decrease,
NE = negligible or no effect, ? = unknown

ATTRIBUTE OF PLANT SPECIES		DEGREE OF ATTRIBUTES	EMBERS	RADIATION	FLAME	WIND
1.	Moisture content of leaves	high/low	?	D/I	D/I	NE
2.	Volatile oil content of leaves	high/low	?	I/D	I/D	NE
3.	Mineral content of leaves	high/low	?	D/I	D/I	NE
4.	Leaf fineness	broad/narrow	?	?	D/I	?
5.	Density of foliage	closely spaced/sparse	?	?	?	D/I
6.	Continuity of plant form	connected/broken	?	?	?	D/I
7.	Height of lowest foliage	high/low	?	?	D/I	I/D
8.	Size of plant – volume	large/small	I/D	I/D	I/D	?
	– spread	wide/narrow	?	?	?	D/I
9.	Dead material on plant	heavy/light	I/D	I/D	I/D	NE
10.	Bark texture	loose/tight	I/D	?	?	NE
11.	Quantity of ground fuel	heavy/light	I/D	I/D	I/D	NE
12.	Particle size of ground fuel	fine/coarse	?	?	I/D	NE
13.	Compactability of ground fuel	packed closely/ loosely	?	D/I	D/I	NE
14.	Mineral content of ground fuel	high/low	?	D/I	D/I	NE

Attack Modification Module

The actual attack experienced by a building will be determined by both the environmental conditions and the landscape. Although the environmental conditions affecting bushfires (windspeed, temperature and humidity) are well known and are incorporated into various fire spread models, their effects on the bushfire attack (burning debris, radiation, flames and wind) on a building are not well defined and there are no models to predict this attack. The attack is also affected by features of the landscape (slope, windbreaks, radiation barriers, cleared areas, etc.) but again no models are available.

The overall attack on a building can be beneficially modified by managing the vegetation, mitigating local winds and changing landscape features. All these measures constitute the 'landscape approach'. More explicitly, this approach involves the selection and location of vegetation, the manipulation of the landform, the siting and design of amenities such as walls, fences, paths, driveways, paved areas, cleared open spaces and the siting of the buildings themselves. A considerable amount of qualitative information and advice on implementing such measures is available and this has been brought together in a recent handbook 'Building in Bushfire-prone Areas – Information and Advice' (Ramsay and Dawkins 1993).

A scheme for assessing the performance of individual plants in terms of their 'flammability', 'ability to produce ground litter' and 'barrier forming ability' based upon their 'attributes' has been developed (Rudolph 1993b). This scheme is summarized in Table 2. To provide an overall assessment for a particular species, the scores could be simply added up, but the result would be distorted as there is, at present, no weighting system for either the attributes or the degree of importance of the three performance characteristics. It should be possible to arrive at an overall assessment using a rules-of-combination method as described by Hopkins (1977). In this method, rules would be developed that state levels at which individual attributes and combinations of them, become critical. In the meantime, certain critical attributes can be red-flagged for special attention. The scheme provides a basis for decisions to be made with regard to managing vegetation.

As indicated above, we can generally only go so far in mitigating the bushfire attack on a building using the landscape approach. In addition, a lack of maintenance of some landscape measures used, especially with regard to vegetation management, can limit the long-term viability of this approach (compared with the building approach).

Building Response Module

In the last ten years much has been learnt concerning the behaviour of building materials subjected to bushfire attack and the influence that design has on the

TABLE 2

Effect of attributes of plants on their performance characteristics.

Effect of attributes on mode of attack
I = increase, D = decrease,
NE = negligible or no effect, ? = unknown
– = effect considered elsewhere in table

ATTRIBUTE OF PLANT SPECIES		DEGREE OF ATTRIBUTES	FLAMMABILITY	PROVISION OF GROUND FUEL	BARRIER FORMING ABILITY
1.	Moisture content of leaves	high/low	D/I	–	NE
2.	Volatile oil content of leaves	high/low	I/D	–	NE
3.	Mineral content of leaves	high/low	D/I	–	NE
4.	Leaf fineness	broad/narrow	D/I	–	I/D
5.	Density of foliage	closely spaced/sparse	I/D	–	I/D
6.	Continuity of plant form	connected/broken	I/D	–	I/D
7.	Height of lowest foliage	high/low	D/I	–	D/I
8.	Size of plant – volume	large/small	I/D	–	–
	– spread	wide/narrow	–	–	I/D
9	Dead material on plant	heavy/light	I/D	–	NE
10.	Bark texture	loose/tight	I/D	–	NE
11.	Quantity of ground fuel	heavy/light	I/D	I/D	NE
12.	Particle size of ground fuel	fine/coarse	I/D	I/D	NE
13.	Compactability of ground fuel	packed closely/ loosely	D/I	D/I	NE
14.	Mineral content of ground fuel	high/low	D/I	D/I	NE

ignition and destruction of buildings. Nearly all the information has been gathered by surveying the fate of buildings in actual bushfires and statistical analysis of the attributes of the landscape and buildings (Cole 1983; Ramsay *et al.* 1986; and Wilson and Ferguson 1986). Such analyses can be used as a model for predicting building survival given certain vegetation attributes and building features, but the one model that has been published (Wilson and Ferguson 1986) is limited in terms of the number of parameters included. Such models are generally specific to the area surveyed rather than of general applicability, and should only be used within the limitations imposed by the original database.

Surveys of buildings involved in bushfires that have been carried out (McArthur and Ramsay, unpublished) since those of the 1983 Ash Wednesday fires (Ramsay *et al.* 1986; Wilson and Ferguson 1986) have tended to support the conclusions of previous surveys. Table 3 summarizes the fate of the houses and indicates the significant role that people played in preventing the destruction of a majority of those houses that survived.

Some experimental work has been carried out to understand the parameters that affect the ignition of timber by burning debris under bushfire conditions (McArthur and Lutton 1991). A laboratory study of the ignition of building details incorporating timber under bushfire conditions, was carried out using 'mock-ups' of the these details using both realistic (leaves and twigs) and artificial miniature timber crib ignition sources. Parameters such as mock-up type, moisture content of the timber comprising the mock-up, temperature and relative humidity of the experimental atmosphere were examined. It was found that all these parameters affected the ignition and flame propagation induced and that under conditions typical of a severe bushfire (40°C, 10 per cent relative humidity, 5 per cent timber moisture content), only very small amounts of fuel were required to cause ignition. These results support the belief, based on field studies, that ignition of exterior building details made of timber plays an important role in the destruction of buildings in bushfires.

A study of the behaviour of aluminium building products in bushfires has been carried out to assess the validity of assertions that they perform poorly under bushfire conditions (McArthur 1991). The work was conducted in two parts: an examination of data from bushfire damage surveys and laboratory experiments in which window assemblies were exposed to radiant heat from a furnace. Data from surveys were augmented with information from a specific questionnaire. Analysis of this data did not provide any support for the belief that houses incorporating aluminium building products are at greater risk of being destroyed. The laboratory experiments on both timber- and aluminium-framed windows showed their performance under simulated bushfire conditions to be similar in that the glazing cracked before the frames became involved.

TABLE 3

Agency carrying out fire fighting activities and fate of houses.
(S = surviving house and D = destroyed house)

	AVOCA AND MELTON VICTORIA		TOCUMWAL NSW		NSW CENTRAL COAST		STRATHBOGIE	
	JANUARY 1985		JANUARY 1990		DECEMBER 1990 AND OCTOBER 1991		DECEMBER 1990	
AGENCY	S	D	S	D	S	D	S	D
Occupants	41	3	16	0	8	0	13	2
Fire brigade	5	5	2	0	6	0	0	0
Occupants and fire brigade	5	0	0	0	2	0	5	0
Others	7	0	0	0	2	0	0	0
None	10	28	1	0	0	5	3	1
Unknown	17	29	9	3	7	7	8	0
Subtotal	85	65	28	3	25	12	29	3
Total	150		31		37		32	

Both the surveys and the laboratory work provide a basis for assessing the response of a building to a given bushfire attack and designing against it using the 'building approach'. The current state-of-the-art with regard to information and advice on measures that might be employed for individual building elements in the building approach has been compiled in a recent handbook (Ramsay and Dawkins 1993). This handbook reproduces the requirements of the Australian Standard which sets out minimum measures for protection against burning debris (Standards Australia 1991), and discusses additional measures which may be used for further protection against burning debris, and measures for protection against attack by radiant heat, flame contact and wind. It is intended that designers will select measures for each building element, tailored to the type of bushfire attack envisaged. However, no quantitative measures are given and expert judgement is required to select appropriate measures.

FUTURE RESEARCH

One can identify the modules of an integrated approach to designing for building survival in bushfires in a qualitative fashion and, with expert opinion, use such an approach to produce a range of design solutions that will undoubtedly improve building survival. However, much more data is required before a quantitative approach can be used. Such an approach is needed if one is to develop appropriate mathematical models of the interface of urban areas with the countryside. This quantitative approach will enable:

(a) trade-offs between landscape and building design approaches;

(b) cost-effective design solutions to be identified; and

(c) actual prediction of building survival.

In particular, data are required on the bushfire attack potential for various plants, how that attack is modified by various landscape components and design measures, and the likelihood that a given attack will cause ignition and destruction of buildings of particular combinations of materials and designs.

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A basis for planning fire to achieve conservation and protection objectives adjacent to the urban interface

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ABSTRACT

The use of planned fire to achieve both protection and conservation objectives in bushland adjacent to developments is a major challenge to land managers. Here we develop and present some simple conceptual models that may help to resolve this problem. By presenting these models we anticipate that they may be further developed in a quantitative manner.

INTRODUCTION

The selection of fire regimes to meet multiple land management objectives remains one of the principal problems facing fire managers and scientists. The problem is perhaps most acute in those lands dedicated for conservation which abut urban, industrial or rural development. Humans are a source of unplanned (UPF) and planned (PF) fires, the latter being employed mainly to alleviate the threat posed to humans by the former. The interaction of these two types of fire will substantially influence both protection and conservation.

Debate is generated by the use of PF to manipulate fuel quantities in order to provide human protection (e.g. McMahon *et al.* 1984). Concurrently there is concern that disturbance regimes in areas managed for conservation have been substantially altered in the recent past (Hobbs and Hopkins 1990) and that some form of PF may be needed for conservation in specific systems (e.g. Saxon 1984). The objective of conservation is to maintain biotic diversity and the processes that are associated with biota (Western 1991). In practice, within discrete areas such as reserves, this translates into maintaining populations of species (i.e. avoiding extinctions).

There are outstanding examples where the use of PF is directly targeted at conservation objectives (Christensen and Maisey 1987). Often strategies for the use of PF are carefully derived to achieve both conservation and protection objectives (Sneeuwjagt 1989). Sometimes, however, conservation is used as an *ad hoc* justification for PF operations directed at protection (Good 1985). Is the sort of PF needed for protection the same sort needed for conservation objectives? How much fire, if any, do we need to inject into an area to conserve biodiversity?

Despite improvements in knowledge of fuels and aids to management in the last 10-15 years, community debate about the management of fire and particularly the use of PF continues. Should there be more PF or less? How much is needed within a landscape to protect an adjacent urban interface? Is a form of PF required for conservation purposes and if so can a single PF operation or program be planned that fulfils both protection and conservation objectives? The answers will depend on the physical location and extent of PF operations in any particular landscape.

Strategies for use of PF in particular areas are often intuitive (Sneeuwjagt 1989), despite the element of sophistication afforded by modern databases, geographic information systems, fire behaviour models and methods for integrating these tools. Formal relationships between the use of planned fires, protection and conservation in landscapes have not been explicitly developed in a quantitative manner. Such an approach may quantify the relationships and assumptions that often lie behind the use of PF in landscapes and assist with the comparison of options.

Here, several conceptual models are presented which summarize ideas that could lead to a quantitative basis for evaluating the contribution of PF to protection and conservation.

We use the Sydney area as a background to illustrate our ideas. There the interface of urban development with bushland is extensive (a scale of $>10^3$ km in length) with much of the bushland composed of fire-prone shrub/woodlands on sandstone soils (Benson and Howell 1990).

THE ROLE OF PLANNED FIRES IN PROTECTION

How effective is PF for hazard reduction in limiting wildfire spread, particularly to an interface? There are few case studies that objectively deal with this question (e.g. Underwood *et al.* 1985; Buckley 1992; McCaw *et al.* 1992). Here we discuss the concept of protection (probably the inverse of hazard) at the interface as a function of the amount (area) of fire prescribed for hazard reduction in adjacent bushland.

For a landscape such as a reserve that has an extensive urban interface, the degree of protection afforded to the interface will be some function of the area subjected to PF for hazard reduction; Figure 1a illustrates the simplest abstract case. We assume that if nearly the whole area is subjected to PF, protection would be close to maximum, whereas if little of the area is treated, protection would be minimal. Between these extremes the size and location of individual PF operations in relation to factors such as fuel type, physiographic features and proximity to the interface will determine the nature of the relationship between protection and total extent of PF. Here we postulate two functions (Fig. 1b) which reflect different and more realistic configurations of PF. If PF is concentrated on aspects with highest fire potential (e.g. steep slopes and high fuel loads), both at the urban interface and in strategic buffers more remote from the interface, it is arguable that the 'efficiency' of protection will be greater (curve I, Fig. 1b); i.e. protection is greater per unit area of land treated. There is resultant low hazard immediately adjacent to the interface.

The lowest efficiency may correspond to scattered distribution of PF operations on inappropriate aspects, low fuel loads and places very remote from the interface, leaving dangerous conditions immediately adjacent to the interface. The last point is important as it presumes that there is some risk of UPF passing through PF areas and reaching the interface. In the case of this second scenario (curve II Fig. 1b), the outcome is that a large area of PF is needed to achieve a high level of protection.

These scenarios can be illustrated informally using an example (Fig. 2) of a severe wildfire in the Brisbane Water National Park, near Gosford (about 80 km north of Sydney) on 23 December 1990. This fire took place under extreme weather (Sydney FFDI = 83, maximum temperature = 40 °C, maximum windspeed = 46 km h^{-1}, relative humidity = 14 per cent) and resulted in the destruction of several houses in the town of Pearl Beach. The path of the fire included a number of areas that had burnt less than 5 years prior to the wildfire (Fig. 2), including the entire valley behind the town (burnt 13 months before). The wildfire spread downhill in this valley. Prior, recent fires probably reduced the level of damage sustained in the town compared to that likely under higher fuel levels.

The outcome of alternatives in this example can be imagined. Given the weather, would more or less protection to the town have been provided by;

(i) a zone of intensively managed fuels, extending several hundred metres around the perimeter of the town;

(ii) a more diffuse distribution of recently burnt patches (but equivalent in area to those in Fig. 2) before the wildfire;

(iii) more extensive (greater area) fuel management by PF remote from the town?

These alternatives would be likely to produce different functions when presented in the form of Figure 1 and thus could be formally compared.

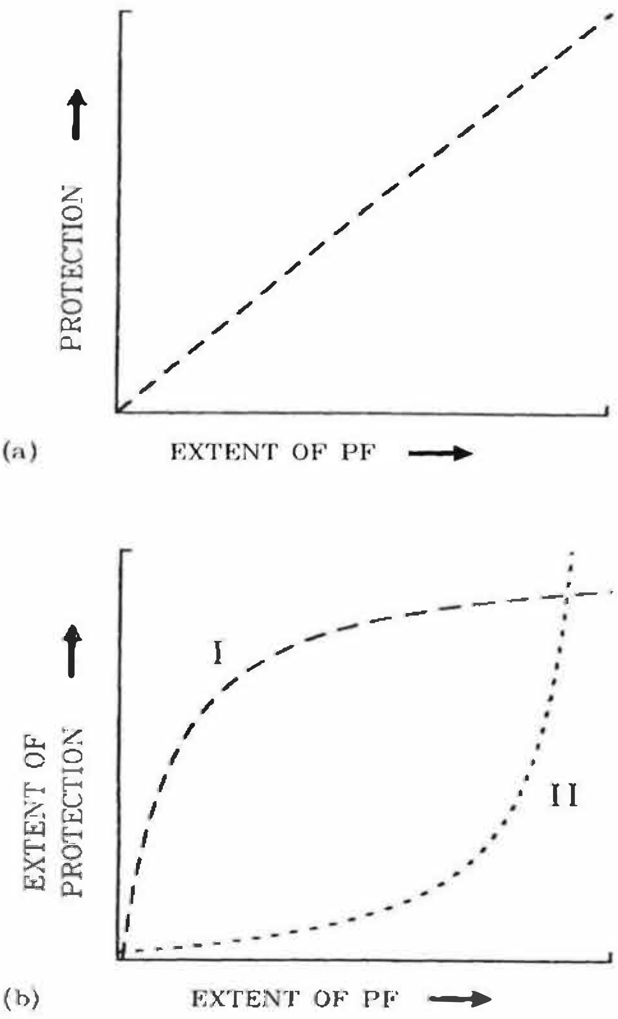

Figure 1. (a) Protection at the interface is presumed to be positively related to the extent of planned fire (PF) in an adjacent landscape. (b) Two forms of this relationship (curves I and II) indicate the sensitivity of the relationship to the location and configuration of areas of PF in the landscape.

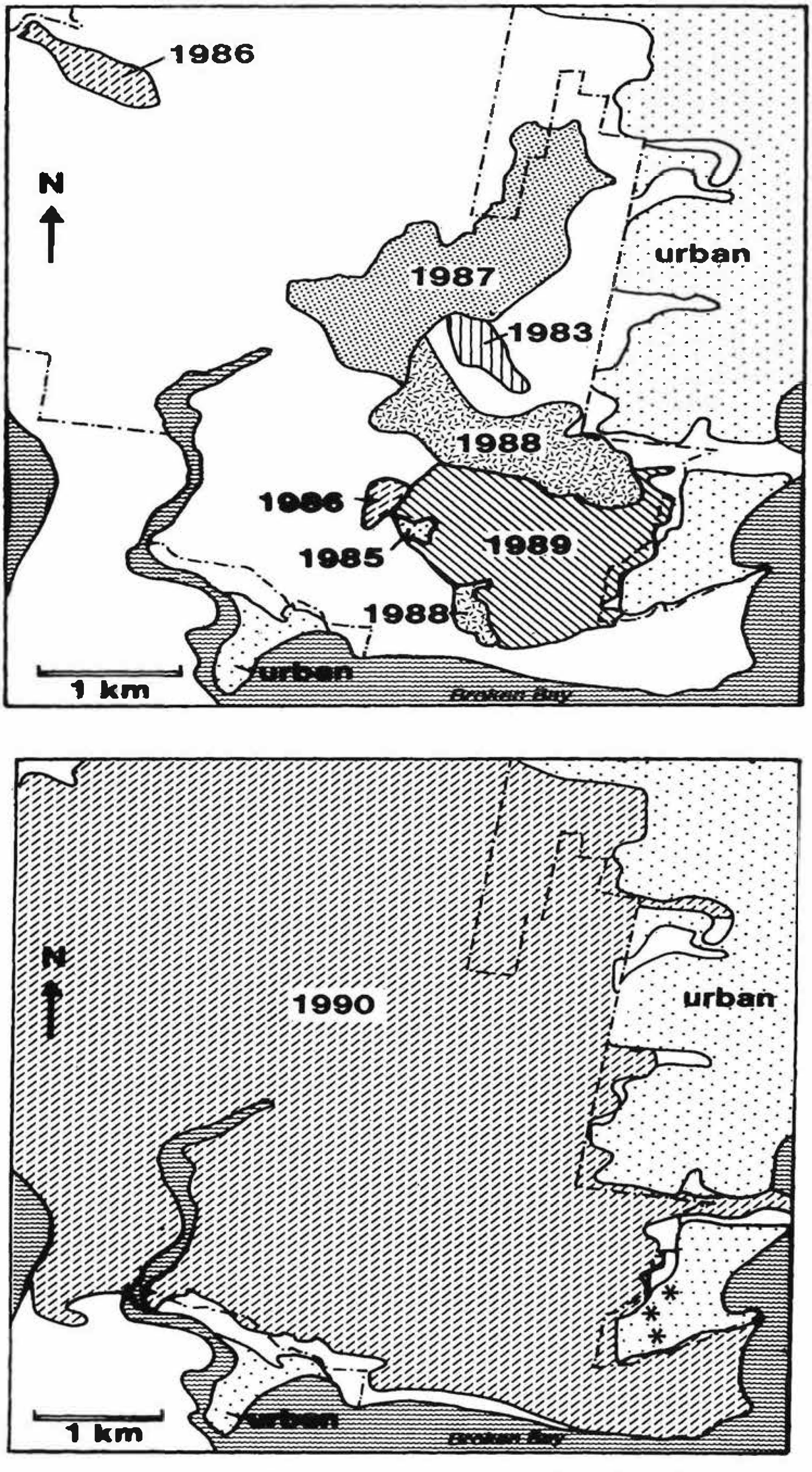

Figure 2. Maps showing pre-fire condition and extent of a major wildfire (December 1990) in the Brisbane Water National Park north of Sydney which resulted in property destruction (denoted by ✱) in the town of Pearl Beach. The direction of headfire spread was from north-north-west.

THE ROLE OF PLANNED FIRE IN CONSERVATION

Specific guidelines and objectives for fire management for conservation are often lacking for particular areas (Good 1981; Press 1989). Management by default concentrates on individual species, usually because they are rare or have been studied. However, this can be detrimental to co-habiting non-target species (e.g. Keith, this volume).

How do we frame guidelines for the management of groups of species? For the plants of fire-prone woodlands and shrublands of the Sydney region, empirical evidence (Cary 1992; Morrison *et al.* 1995) suggests that there is a relationship between fire regime variability and the number of species in a site (Fig. 3). Variability of fire frequency is of particular importance in determining floristic diversity in this flora (Cary 1992; Keith and Bradstock 1994; Morrison *et al.* 1995).

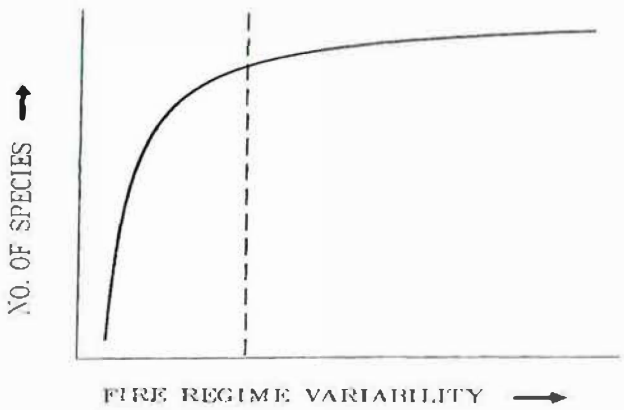

Figure 3. Number of plant species at a site level is postulated to be a function of fire regime variability in fire-prone shrub/woodlands of Sydney. Fire regime variability exceeding a threshold (dashed line), will maintain species numbers at or near their maximum.

Given the importance of fire frequency we would define a non-variable or uniform regime as consisting of successive fires spaced identically in time. In the shrubland and woodland communities around Sydney the evidence suggests that such a uniform fire frequency (low variance in frequency) may result in the eventual elimination of species irrespective of whether the regime is composed of short (<10 years), moderate (10-30 years) or long (>30 years) intervals between fire. Note that each fire frequency scenario affects species in different categories according to life history, though the net result in each case is similar: i.e. a reduction in number of species. It follows that to maintain all species, a mixture of fire intervals of different length (high fire frequency variance) is required, including the occasional incidence of a very short fire interval. The mechanisms behind this are more fully explained by Keith (this volume) and Keith and Bradstock (1994). It is important to note that these conclusions apply at a fine level of scale (e.g. in stands and sites).

From Figure 3 it is possible to define a threshold for fire regime variability which demarcates different levels of species presence at sites. Fire-management to maintain high numbers of species at a site would therefore be aimed at keeping fire regime variability above this threshold. Note that we postulate that a wide range of fire regime variability (or permutations of intervals between fire) is compatible with maximum or near maximum species presence. Definitions of these fire regime thresholds for some Sydney sandstone communities are provided in Bradstock *et al.* (1995).

IMPORTANCE OF FIRE FREQUENCY

Given that it is possible to define the level of fire regime variability needed to maintain plant species at a fine scale, what role do planned fires play in contributing to this variability across a landscape? It is likely that fire regime variability at this level of scale will be some function of the area covered by planned fires in that landscape.

A model is presented in Figure 4 which relates the spatial extent of fire regime variability (e.g. the proportion of sites in which species are maintained close to the maximum) to extent of planned fires. The landscape is assumed to be composed of a matrix of many sites. This assumes that a regime of UPF prevails in the landscape, the nature of the model being highly dependent on the characteristics of UPF and most importantly the interaction or overlap between PF and UPF at particular sites.

We predict (Fig. 4a) that when PF extent is low then UPF is predominant in the landscape. A proportion of sites will therefore be subjected to relatively uniform fire regimes and species presence in those sites will be reduced. As the extent of PF increases, an increased proportion of sites (greater area) will be subject to fire regimes with sufficient variability to maintain species numbers close to the maximum. This comes about in several ways through the interaction of UPF and PF. Unplanned fires in extreme conditions will burn through areas subject to PF (e.g. Fig. 2). Such areas are therefore subject to a high chance of a short interval between fires.

Conversely in some situations the presence of a PF will limit the extent of an UPF by moderating its spread rate and extent. Thus UPF will not reach some sites, changing the fire frequency in those sites to a more variable state than would have prevailed without PF. The model postulates that there is a level of PF extent that optimizes this overlap effect and thus optimizes fire regime variability across a maximum number of sites in the landscape. The model will be sensitive to the locality of PF in a landscape as well as extent of PF (Fig. 4b). For example, there are often defined wildfire paths in particular parcels of land according to patterns of ignition and weather (Minnich 1989). The placement of PF in relation to these paths will determine overlap effects.

We postulate that as PF extent increases beyond the optimum level either uniform PF regimes may become predominant, or the overlap between PF and UPF may become too extensive and repetitous through time, resulting in fire regime uniformity over an increasing proportion of sites.

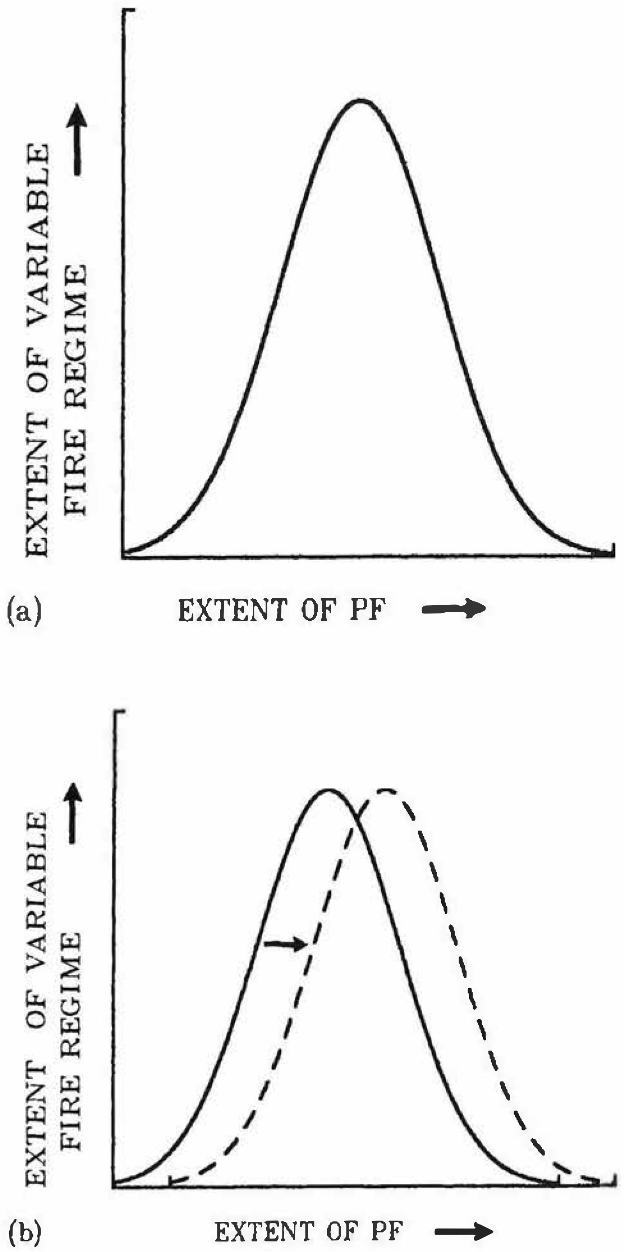

Figure 4. (a) For a landscape, fire regime variability sufficient to maintain high numbers of plant species at sites, will be some function of the extent of PF. (b) The relationship presumes that a particular regime of unplanned fire (UPF) prevails in the landscape. The effect of a shift in UPF (broken line) on that relationship is shown.

A MODEL TO EVALUATE THE DUAL ROLES OF PLANNED FIRE

We have suggested how the contribution of PF to protection and conservation objectives can be formally evaluated on a common basis (areal extent of PF). If we are to know whether the sort of PF needed for protection is the same as that needed for conservation, we need to merge the separate strands developed above. Such a model is given in Figure 5 illustrating different scenarios. We present these as alternatives which may represent the situation in a landscape at some point in time. The most important feature of these models is the relativity of the pairs of curves with respect to the horizontal axis.

The first example (Fig. 5a) illustrates a situation where an optimal solution for both objectives is difficult to achieve. If protection is pursued as a priority then conservation will be sub-optimal (PF extent will exceed the conservation optimum). In the second example (Fig. 5b) the PF optimum for both objectives coincides. Note here that there is a fairly wide range of PF values that produce near optimum solutions for both objectives. In the third example (Fig. 5c) both objectives coincide but protection is achieved more efficiently. We may be tempted to earmark this as the ideal management scenario for an area. A variant is given by the fourth example (Fig. 5d), notable because protection is maximized at a lower level of PF extent than conservation (the inverse of the first scenario) with the implication that further PF can be targeted solely to achieve conservation.

We could define a sub-optimal level of fire regime variability in the landscape as being an acceptable conservation objective provided that it is sufficient to conserve species within that landscape. Irrespective of the level of fire regime variability that is acceptable in terms of conservation, this approach allows us to think about how much PF will contribute to that variability.

Undoubtedly, further examples could be generated and their sensitivity to factors such as change in UPF paths and location of PF could be explored. At this stage we stress the value of using this approach to look clearly and objectively at the consequences of management actions. Which example applies to a particular parcel of land? How will it vary over time as PF is used? How can the use of PF be adjusted to produce a better outcome for both objectives? The answers to these questions beg quantitative solutions to this model.

Many of the tools for producing a quantitative solution are available. For example, fire spread models in conjunction with data on terrain and fuels can be used to produce scenarios from which a quantitative relationship between area of PF and protection (e.g. Fig. 1) can be developed for a given parcel of land, presuming certain wildfire scenarios. Some aspects of the problem, however, require further work. In particular, we need to know more about:

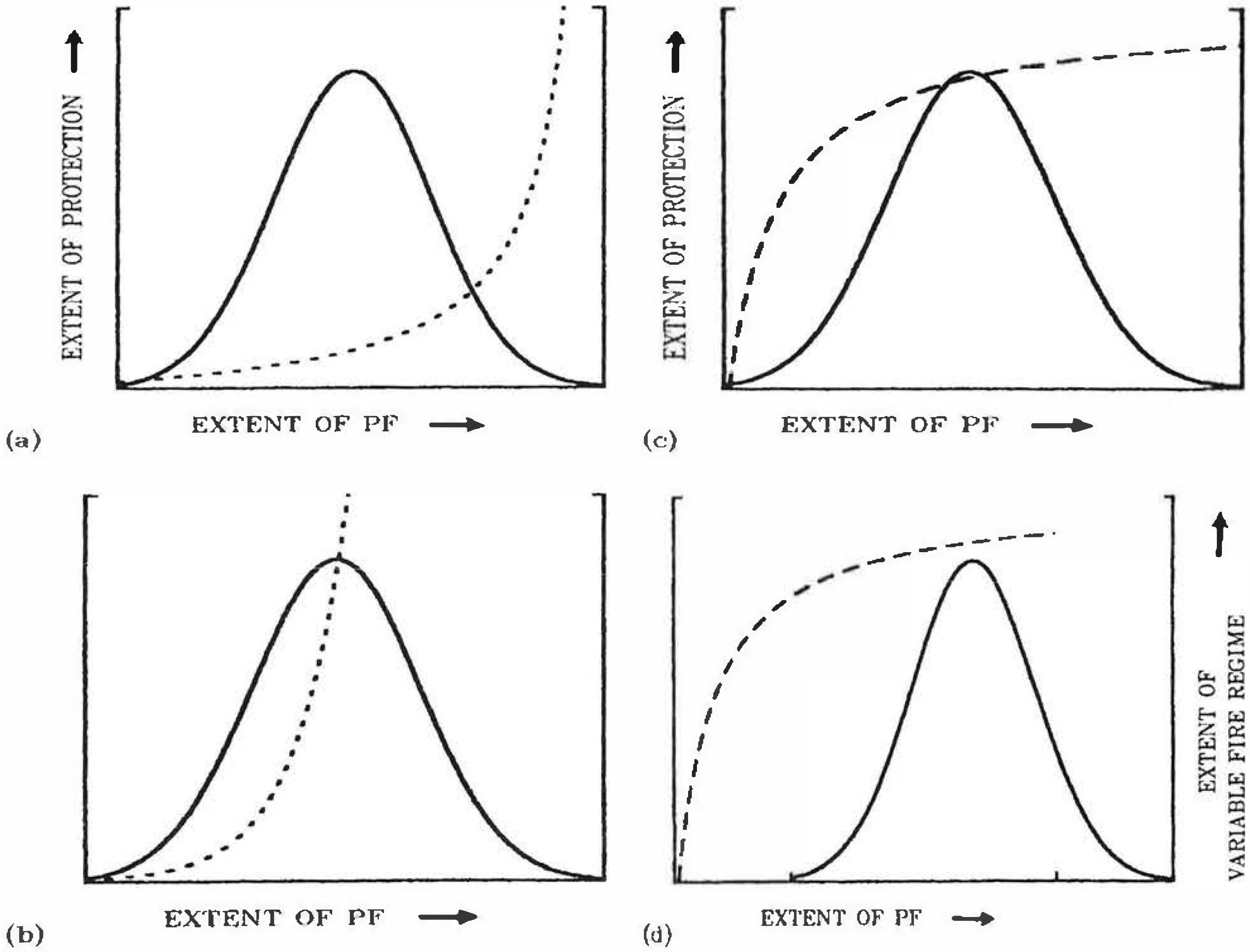

Figure 5. A model with examples (a,b,c,d) depicting different relationships between PF extent and conservation (solid line) and protection (dashed line).

(1) the definition of fire regime thresholds for local or 'site' scale extinction of functional groups of species that correspond with identifiable vegetation communities;

(2) the interaction between PF and UPF as a determinant of UPF spread.

The latter element is important and, as noted, is an understudied part of fire science. The issue of overlapping fires, particularly at short intervals, is pivotal.

Underpinning the use of PF in many conservation and protection programs is the assumption that PF interacts with UPF to truncate UPF spread, intensity and extent. While the injection of planned fires will undoubtedly alter the nature and extent of unplanned fires, the actual limits of such an effect are largely unknown for many systems.

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Fire protection and the urban/rural interface 'Constraints or Opportunities' (a discussion paper) (abstract only)

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ABSTRACT

Fire protection and safety at the urban/rural interface throughout Australia has been subject to considerable research, with findings tending to focus on the more technical aspects of standards, specifications, planning processes, environmental considerations and fire suppression. Internationally, the same aspects have been analysed, researched and documented but little reference has been made to equally relevant issues that are an integral aspect of ensuring fire protection at the urban/rural interface.

This discussion paper sets out to highlight the importance of social, funding and educational issues in ensuring fire safety at the urban/rural interface and seeks to analyse their influence in relation to more orthodox technical issues. It is argued that developers, planners, State and Local Authorities, fire agencies, land managers and other stakeholders should be more aware of social, funding and educational issues and should consider them more comprehensively when determining planning, development and maintenance cycles at the urban/rural interface.

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Improve people's responses to bushfire threats by addressing their motivations (abstract only)

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ABSTRACT

Many people do not make their houses bushfire safe; they do not learn simple survival strategies about sheltering in houses or vehicles; they do not evacuate well before a fire arrives nor by the safest route; they don't keep out of fire areas; or they unnecessarily attempt to fight the fire. This inappropriate behaviour is a major reason that bushfires kill people and destroy houses. Bushfire and other emergency agencies aim to reduce the losses of life and property partly by influencing these behaviours.

Behaviour can be changed, but only if motivation is changed. Providing information and advice (e.g. distributing educational materials before fires on safe houses and the need for any evacuation to be early) and instructions (e.g. telling people during fires when and how to evacuate) in itself causes no change in behaviour. People's perception of personal risk, convenience, needs, priorities and relevance of information will strongly affect their motivations and actions, as illustrated by observations that people:

- allow their lives to be threatened by such factors as an unroadworthy car, poor diet, lack of exercise and continued use of cigarettes. These factors generally have a higher risk and profile than bushfires but have also been difficult to change.
- respond to a general fire threat (e.g. that there is a fire somewhere in their district) by trying to obtain better information, so they can choose a good strategy. The alternative of immediate action could cause them unwarranted inconvenience (e.g. being late for an important appointment despite not actually being in danger), or a bad strategy (e.g. driving towards rather than away from the path of the fire).
- are highly motivated during a fire to protect family members or possessions, if necessary by driving along a possibly dangerous route to reach them.

People fail to address many priorities that may to others appear legitimate; and do address many other priorities besides personal survival.

Professional conditioning may lead fire managers to identify readily these responses as being inappropriate but not to appreciate how the general population reacts to unfamiliar circumstances. Fire managers may benefit from contemplating their own motivations and focus in non-fire situations, by participating in the following exercise. Picture travelling to an important wedding and hearing of some roads being cut by rising flood-waters: do you necessarily turn back or do you seek more information or take some measured chances? Later, while ankle-deep in the rising waters and fading light, with your very best friends stuck nearby in deeper water and calling for your help, do you simply head for dry land without risking any time and effort to help them?

The more that bushfire and emergency agencies recognize and work with people's motivations the more that people will respond to fire threats effectively. Incisive studies of human behaviour before and during bushfires, and of the literature on community psychology in response to other types of disaster, should help identify people's behaviour patterns and strategies for influencing them. Outcomes may include efforts to improve the processes which allow community members to network high quality information during fire events; and to develop leadership throughout the community, to increase the extent of effective decision-making during fires.

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Atmospheric trace gas emissions from tropical Australian savanna fires

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ABSTRACT

Frequent and often annual fires are a major feature of the savannas of northern Australia, but the role of emissions from these fires in atmospheric chemistry remains poorly quantified. The fuel of these fires is normally grass and leaf litter. Fire intensities are relatively low and crown fires are rare. In this paper, the results are presented of measurements of the prompt release of trace gases from savanna fires at Kapalga, NT. The emission ratios of carbon species from the Kapalga fires were similar to those measured in savannas elsewhere in the world. The production of photochemical smog and hence tropospheric ozone is probably one of the main undesirable impacts of these fires on the atmosphere. The effects of possible fire management options on the trace gas emissions were considered by simple modelling of the dynamics of fuel loads and fire intensities. Reducing the fire frequency would increase the likelihood of fire intensities being sufficiently high to reduce the tree cover, with a consequent net release of carbon and nitrogen into the atmosphere. The release of trace gases in the absence of fires needs to be quantified to enable more complete modelling of consequences for the atmosphere of different management regimes.

INTRODUCTION

Biomass burning, especially in the tropics, is now considered to be a globally significant source of atmospheric trace gases (e.g. CO and CH_4) (Crutzen and Andreae 1990), yet at present the global emissions of trace gases from biomass fires are poorly quantified. Until more comprehensive data on trace gas emissions, burning efficiencies, fuel loads and fire frequencies are compiled, the impacts of biomass burning on the atmosphere cannot be assessed accurately. Most fires in the tropics are lit by humans, and this information would help provide a rational basis for modifying existing fire regimes in order to alter the release of trace gases.

In the northern sub-coastal zone of the Northern Territory, fires typically occur in two years out of three (Braithwaite and Estbergs 1985). This frequency of fires is among the highest in the world. Although the fuel loads are relatively low compared with biomass fuels in other vegetation types such as temperate Australian forests (Walker *et al.* 1986), the average amount of fuel consumed *per year* is higher in savannas because of the extremely high fire frequency. Hence the pyrogenic release of carbon and nitrogen gases per year will be higher. In this paper, we report measurements of the prompt release of several important trace gases during fires in the savannas of Kakadu National Park in the Northern Territory. Preliminary data are also presented on the changes in fuel loads and fire intensities with increasing period without fire. The implications of these data for manipulating fire regimes to minimize the emissions of greenhouse gases are discussed.

MATERIALS AND METHODS

Study Site

Kapalga Research Station is 180 km east of Darwin and within Kakadu National Park. The upland ridge between the South Alligator and West Alligator Rivers has been divided into management compartments which represent the catchments of intermittent streams. Each compartment is 15-20 km^2 in area and surrounded by fire breaks. The dominant vegetation of the upland ridge of Kapalga is open forest of *Eucalyptus miniata* and *E. tetrodonta* with the understorey dominated by the annual grass *Sorghum intrans* and perennial grasses such as *Heteropogon triticeus*.

Four experimental fire regimes have been imposed on thirteen compartments on Kapalga. The fire regimes are as follows:

1. 'Early dry season annual'; burnt each year during the early dry season (June). (Compartments E, K, and P.)

2. 'Progressive'; burnt three times during the dry season each year (June, July and September). (Compartments A, B, and H.)

3. 'Late dry season'; burnt each year during the late dry season (September). (Compartments F, G, and L.)

4. 'Natural'; unburnt control; fire is actively excluded from these compartments. (Compartments C, M, Q, and S.)

The first experimental fires were imposed in 1990. Fire had been excluded from the compartments for either two or three years before 1990.

Smoke Sampling

Smoke samples were collected from the ground in June and September 1991 using evacuated 0.6 L glass flasks. The type of combustion that occurred at the point of sampling was classified as flaming or smouldering. Samples of 'clean' air were collected upwind of each fire site for the determination of the background ***mixing ratios*** of each trace gas. The mixing ratio is defined as the ratio of the mass of a given gas to that of the remaining gas in a mixture. In September 1991 and September 1992, samples of smoke were collected at low altitude (50-700 m above ground) from an aircraft as it traversed fresh smoke plumes. Samples of 'clean' air were also collected a short distance upwind of the fire sites for the determination of background mixing ratios. Further details of the methods for sampling the smoke are presented by Hurst *et al.* (1994).

The ***excess mixing ratios*** were calculated as the difference between the background mixing ratios for each gas and the respective mixing ratios in the smoke samples. The excess mixing ratios for three carbon species are designated in the text as DCO_2, DCO, and DCH_4. The ***emission ratios*** for carbon monoxide and methane were defined as the ratios between the excess mixing ratios of these gases and that of carbon dioxide. That is DCO/DCO_2 and DCH_4/DCO_2 respectively.

The concentrations of carbon species were measured in the samples using a Fourier-transform infrared (FTIR) spectrometer and matrix-isolation FTIR spectroscopy (Hurst *et al.* 1994). The nitrogen species NO, NO_x and NH_3 were measured using a chemiluminescence analyser (Hurst *et al.* 1994).

Fuel Loads

Fuel loads were measured each year since 1990 in permanent 20 m *x* 50 m plots situated within the ***E. miniata, E. tetrodonta*** open forest on each compartment. The fuel load in each plot was estimated annually as the mean of the herbaceous biomass and litter within eight randomly placed 0.25 m² quadrats. The measured fuel loads represented sites with a range of periods without fire from 1 to 6 years. Sites where fire has been excluded for more than two years are rare in this region, and these data represent some of the only available measurements of fuel loads under these conditions.

RESULTS

Emission Ratios of Carbon Species and Combustion Efficiencies

No significant differences in the emissions of trace gases were detected between the samples collected during the June fires and those in September. Therefore the data were bulked irrespective of time of sampling. In Table 1 is shown the mean emission ratios for carbon monoxide and methane for ground-based smoke samples collected from flaming and smouldering fires at Kapalga and for aircraft-based samples from fires at Kapalga. These values are compared with measured values from savanna fires in Côte d'Ivoire and Brazil, and from forest fires in the Sydney region.

The mean carbon monoxide and methane emission ratios of ground-based smoke samples were lower for flaming fires than for smouldering fires in both savanna fires at Kapalga and forest fires near Sydney (Table 1). While the emission ratios were similar for flaming fires in both the savanna and the forest, the values for smouldering fires were lower for the savanna than the forest (Table 1). This indicates that the combustion efficiencies were greater for flaming fires than smouldering fires generally, but that smouldering fires in the savanna had a greater combustion efficiency than in the forest.

The carbon monoxide and methane emission ratios for aircraft-based samples from the savanna fires at Kapalga were in good agreement with those measured from savanna fires in Brazil and Côte d'Ivoire. The values for the aircraft-based samples were between those for ground-based samples of smoke from flaming and smouldering fires in both Sydney and Kapalga (Table 1). The values were, however, higher for forest fires than for savanna fires, indicating a greater overall combustion efficiency for savanna fires. It was calculated from these values that between 69 per cent and 74 per cent of the carbon consumed by the savanna fires at Kapalga is released by flaming combustion compared with between 52 per cent and 57 per cent in the forest fire (Table 1).

Partitioning of Carbon and Nitrogen During the Fires.

The partitioning of the fuel carbon and nitrogen during the fires can be estimated by calculating the ***emission factors*** of each gaseous species and determining the amounts of carbon and nitrogen which remained in the ash after combustion. These were calculated as described by Hurst *et al.* (1984). The partitioning of carbon and nitrogen during the fires is shown as pie charts in Figures 1 and 2 respectively.

TABLE 1

The DCO/DCO2 and DCH4/DCO2 emission ratios of smoke samples from savanna fires at Kapalga, compared with the ratios from forest fires in the Sydney region and savanna fires in Côte d'Ivoire and Brazil. The percentage contribution of flaming combustion to the overall combustion was calculated from the values. Numbers in parentheses indicate standard deviations.

SITE	MEAN EMISSION RATIO			C RELEASE BY FLAMING COMBUSTION (%)
	FLAMING FIRE (GROUND-BASED)	SMOULDERING FIRE (GROUND-BASED)	TOTAL FIRE (AIRCRAFT-BASED)	
DCO/DCO_2				
Kapalga	0.055 (0.023)	0.168 (0.027)	0.090 (0.026)	69
Sydney[a]	0.05	0.175	0.11	52
Brazil & Côte d'Ivoire[b]	-	-	0.053 - 0.113	-
DCH_4/DCO_2				
Kapalga	0.0021 (0.0015)	0.0101 (0.0039)	0.0042 (0.0019)	74
Sydney[a]	0.0022	0.013	0.0068	57
Côte d'Ivoire[c]	-	-	0.0038 (0.0011)	

[a] D. Griffith, unpublished data.

[b] Greenberg *et al.* 1984; Bonsang *et al.* 1991; Lobert *et al.* 1991; Ward *et al.* 1992.

[c] Delmas *et al.* 1991.

The combined emissions of CO_2, CO, CH_4, total non-methane hydrocarbons (NMHC) and particulate carbon (PC) approximates all carbon-containing emissions. Carbon emissions were dominated by CO_2 which represented 86.8±2.8 per cent (± standard deviation) of the fuel carbon, while CO accounted for 7.8±2.3 per cent (Fig. 1). Emissions of total NMHC and CH_4 comprised less than 1 per cent each of the fuel carbon.

The nitrogen-containing emissions could only be partitioned to NO_x, NH_3, N_2O, HCN, CH_3CN and ash. The measured nitrogen emissions were dominated by NO_x (21±8 per cent of fuel N) and NH_3 (23±13 per cent) with N_2O, HCN, and CH_3CN each representing less than 1 per cent of the fuel nitrogen. The emission factors for NO_x, NH_3, N_2O, HCN, and CH_3CN together accounted for only 46±15 per cent of the fuel nitrogen. The nitrogen contained in the ash represented 11±5 per cent of the fuel nitrogen, so in total, our measurements accounted for only 56±16 per cent of the fuel nitrogen. This shortfall in detected nitrogen-containing emissions from biomass burning is now well documented, and recent experimental evidence indicates that 33±13 per cent of fuel nitrogen is emitted as N_2 during flaming combustion (Kuhlbusch *et al.* 1991). Any remainder of the undetected fuel nitrogen may have been emitted as higher molecular weight compounds.

Fire Behaviour and Fuel Consumption Under Different Fire Management Options

The mean biomass of fuel and fire intensity of plots on Kapalga with varying period without fire is presented in Table 2. Fuel loads were least under annual fires, but there was no evidence that fuel continued to accumulate after about two to three years without a fire. The available evidence indicates that fire intensities tend to be greater on sites that have not been burnt for two or more years than on annually burnt sites. Fire intensities also tended to be greater for late dry season fires than early (see Williams, this volume).

DISCUSSION

The release of trace gases from fires in the savannas at Kapalga appears to be similar to that from savannas in other parts of the world (Table 1). A characteristic feature of savannas is the prevalence of grassy fuels, in contrast to greater importance of woody fuels in forest communities. The dominance of flaming combustion and the greater overall combustion efficiency of the savanna fires compared with forest fires in southern Australia (Table 1) is consistent with these differences in fuel types.

TABLE 2

The measured fuel load and fire intensity of experimental compartments at Kapalga with varying periods without fire in the years from 1990-1993.

PERIOD WITHOUT FIRE (Y)	COMPARTMENTS	YEAR	FUEL LOAD (T HA⁻¹)*	MEAN FIRE INTENSITY ** (kW m⁻¹ X 10³) EARLY	LATE
1	A, E, F, G, H, & P	1991, 1992, & 1993	5.1 (1.4)[a]	1.7 (n = 10)	5.2 (n = 4)
2	F & P	1990	7.8 (1.7)[b]	10 (n = 1)	-
3	A, C, E, G, H, & M	1990	10.3 (0.9)[c]	10 (n = 1)	15 (n = 1)
4	C & M	1991	8.1 (1.0)[bc]	-	-
5	C & M	1992	8.5 (0.2)[bc]	-	-
6	C & M	1993	6.8 (1.7)[ab]	-	-

* different letters indicate significant differences ($p < 0.05$).

** A.M. Gill, R.J. Williams and P. Moore: unpublished data; data were not available for all fires.

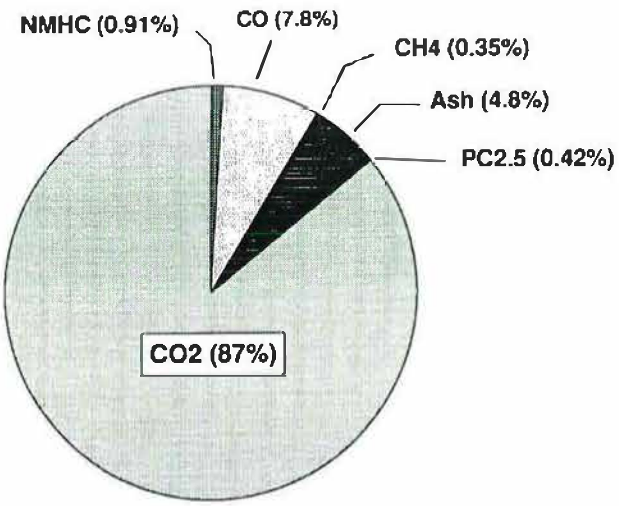

Figure 1. Partitioning of fuel carbon during savanna fires at Kapalga. PC 2.5 is emitted particulate carbon (<2.5 mm diameter, see Hurst et al. *1994).*

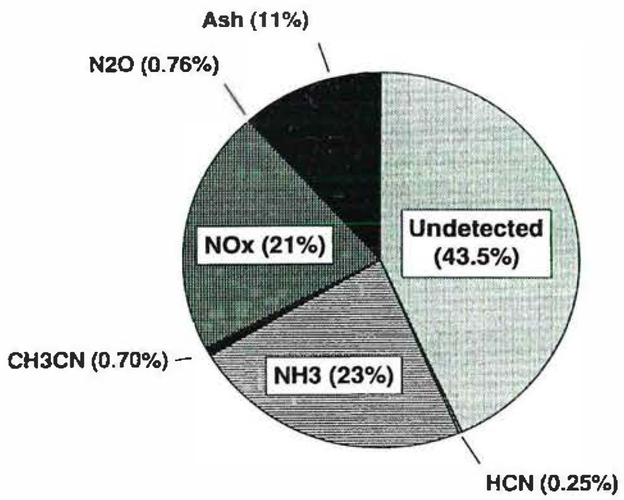

Figure 2. Partitioning of fuel nitrogen during savanna fires at Kapalga.

Carbon dioxide, which was the main carbon species released during the fires at Kapalga (Fig. 1), is a major greenhouse gas in the atmosphere. However, if the structure and productivity of the savanna vegetation is not changing over time, then the growth of vegetation in the weeks and months after each fire should take up similar amounts of carbon to that released. Thus, the net release of carbon to the atmosphere should be close to zero in this region because there is little evidence of substantial changes in vegetation structure of savannas in the region in response to fire (Bowman *et al.* 1988; Lonsdale and Braithwaite 1991). Nevertheless, there are concerns that present fire regimes are limiting the recruitment of woody species which may lead to reduced tree cover (Duff and Braithwaite 1989). If this is occurring, then the result would be a positive net release of carbon and possibly a reduced capacity to absorb and store carbon.

The other greenhouse gases released during the fires, methane and nitrous oxide, were both released in small amounts (Figs 1 and 2). It has been estimated that the methane released from biomass burning in Australian savannas represents about 10 per cent of all methane sources in Australia (Hurst *et al.* 1994). In contrast, the release of nitrous oxide by savanna fires represents less than 3 per cent of other natural and anthropogenic sources.

The gases CO, CH_4, NMHC, and NO_x which were released in significant quantities by the fires at Kapalga (Figs 1 and 2) are precursors to the formation of photochemical smog. The production of photochemical smog from these gases is recognized as an important consequence of biomass burning in Africa and South America (Cros *et al.* 1991). Photochemical smog in turn, leads to the formation of tropospheric ozone, which is a very effective, albeit short-lived, greenhouse gas. Although tropospheric ozone was not measured as part of this study, its formation is likely to be one of the main undesirable consequences of the gaseous emissions from the fires in Australia's northern savannas.

If one wishes to manage the savannas to minimize the production of greenhouse gases, it cannot be achieved by simply reducing fire frequency from almost annual to one in two or one in three years. Although the *per year* consumption of grassy fuel by fire would decrease if fire frequency were reduced, the intensity of fires would tend to increase because the fuel loads are greater (Table 2). As well, there is no evidence that soil organic matter levels increase when fires are excluded from these savannas (Cook, unpublished data). With greater fire intensities tree death is likely to increase (see Williams, this volume). As a consequence, the carbon and nitrogen which had been stored in the tree stems would be released to the atmosphere due to decay and combustion over several years. Thus a positive net release of carbon and nitrogen could result from reducing fire frequency if this is accompanied by increased death of trees. As well, proportionately more methane would be released because the stems of the trees would burn by smouldering fires rather than by flaming fires (see Table 1).

The fate of the 'fuel' in the absence of fire must also be accounted for in modelling the effects of different fire regime on trace gas emissions. Fuel loads at Kapalga did not continue to increase after three years without fire (Table 2). This indicates that biological oxidation of organic matter is of major importance in these savannas. The emissions of trace gases during the biological oxidation of the 'fuel' are more difficult to measure than pyrogenic emissions because of their greater spatial and temporal variability. Some evidence suggests that biological oxidation may release more greenhouse gases than pyrogenic oxidation. Termites, for example, release about 1.2 per cent of the carbon they consume as methane (Khalil *et al.* 1990; Holt 1991) compared with only 0.35 per cent for fires at Kapalga (Fig. 2). As well, termites may release a greater proportion of nitrous oxide per unit of biomass consumed than fires. Further work is required to quantify these biogenic emissions.

CONCLUSIONS

The release of trace gases from fires at Kapalga was similar to that from savannas elsewhere in the world. The net release of carbon and nitrogen is likely to be close to zero under present fire regimes because the structure and productivity of these savannas does not appear to be changing. The main undesirable effect of these fires on the atmosphere is likely to be the production of photochemical smog and hence tropospheric ozone. It may be possible to change the present fire regimes to minimize the production of precursors to photochemical smog. If fire frequencies were reduced substantially, the biological oxidation of unburnt biomass may release equivalent or even greater quantities of trace gases than those from fires, but would also predispose the savannas to more intense fires resulting in greater tree death. Maintaining a regime of frequent but low intensity and patchy fires is probably the best practical way to minimize the release of undesirable trace gases.

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Contribution by bushfire smoke to photochemical smog

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ABSTRACT

Visibility reductions in the Perth region, due to smoke from bushfires and controlled burns, have at times been measured by the Western Australian Department of Environmental Protection to be in excess of accepted standards. The smoke has been observed to include high concentrations of hydrocarbons, with the potential to contribute to the formation of photochemical smog.

During the spring and summer of 1992-1993, two photochemical smog events occurred which were clearly associated with the passage of smoke from controlled burns. The background to these events, and some implications, are discussed.

INTRODUCTION

Since the beginning of 1990, the Department of Environmental Protection of Western Australia (subsequently termed the DEP) has monitored haze events in the Perth region using nephelometers. Instruments have been located at the DEP's Caversham, Queen's Buildings (Perth) and Hope Valley sites. (See Fig. 1)

Two major classes of visibility impairment have been identified. One occurs during winter nights, and appears to be due to the accumulation of smoke from domestic fires. The other occurs during early summer evenings, with a moderate sea breeze still present.

A major source of the latter class of episode was found to be controlled burns conducted by the State's Department of Conservation and Land Management. The need to manage the impact of smoke from controlled burns has since been accepted by that department. Their summer burn program now includes enhanced smoke management methods, drawing heavily on specialized forecasting provided by the Bureau of Meteorology.

While the most immediately apparent effect of the smoke was on visibility, chemical contaminants were also present. At the Hope Valley site, levels of both methane and other hydrocarbons have been measured. (The latter are conventionally termed non-methane hydrocarbons, abbreviated NMHC.)

During the summer smoke episodes, non-methane hydrocarbon levels were observed to rise to extreme levels, in almost exact time correspondence with the scattering coefficient measured by the nephelometer. Collected smoke samples were analysed using chromatography, but the analysis showed a mass of peaks, with no clearly identifiable species.

The presence of hydrocarbons at such concentrations was of concern to the DEP, because of their potential to contribute to the formation of photochemical smog, and because some smoke episodes were seen to extend through the morning of the following day.

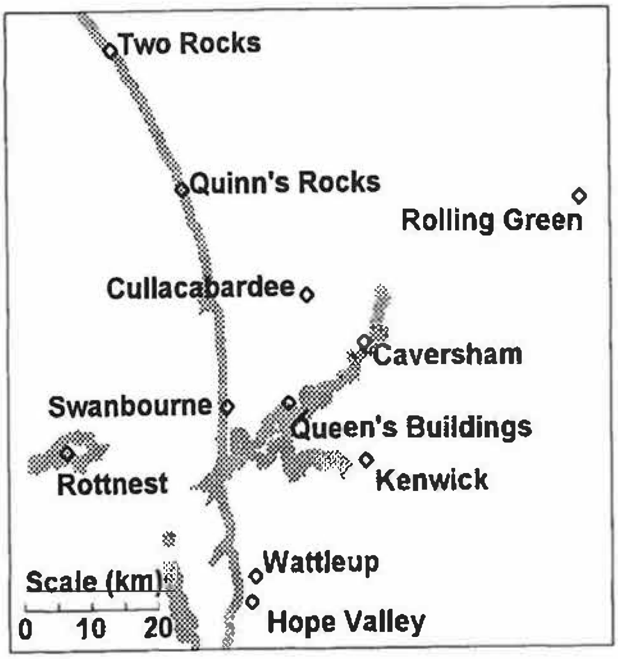

Figure 1. Air quality monitoring sites currently operated by the DEP. Sites other than Hope Valley, Wattleup, Queen's Buildings and Caversham are components of the Perth Photochemical Smog Study.

PHOTOCHEMICAL SMOG PROCESSES

The term 'photochemical smog' refers to a combination of chemical species in the atmosphere, which is the product of reactions involving initially a mix of nitrogen oxides and hydrocarbons with normal atmospheric gases.

The principal components of concern are ozone and a range of oxidated nitrogen species, which are bronchial irritants and corrosive to varying extents. The maximum hourly average concentration for ozone considered acceptable by the United States Environmental Protection Agency and Victorian Environmental Protection Authority is currently 120 ppb (parts per billion). However, as a result of recent epidemiological studies, the recommended hourly average limit for ozone has decreased, with most recommendations in the 75-90 ppb range. A review of these is given by Streeton (1990).

Ozone is produced naturally in the atmosphere by the reactions

$$NO_2 + h\nu \rightarrow NO + O^*$$

$$O^* + o_2 \rightarrow o_3$$

where 'h' represents the energy of a photon of ultraviolet light, and O* indicates an oxygen atom in an excited state. The NO produced is re-oxidized slowly in the clean atmosphere to NO_2, maintaining an equilibrium that leads to a low background level of O_3 of about 15-25 ppb.

The rate of NO_2 regeneration, and consequently ozone production, is considerably accelerated by the presence of reactive volatile organic compounds ('VOC'), and by the presence of nitrogen oxides from anthropogenic sources (i.e., those originating from human activities). In these circumstances, the progress of photochemical smog reactions in strong sunlight leads to a significant increase of ozone concentrations, peaking typically in the mid-afternoon period.

Emissions inventories generally show that the major sources of volatile organic compounds are motor vehicles and industry, but that biogenic emissions - those from vegetation - are also significant. Carnovale *et al.* (1991) estimated that for average Melbourne summer temperatures around 25°C, about 10 per cent of total emissions arose from this source, but that emissions may rise by 60 per cent for a temperature increase from 25 to 30°C. The VOC's emitted from vegetation tended to be of high reactivity, further enhancing their influence.

OBSERVATIONS

Figure 2 shows an example of an overnight smoke episode, in which both visibility and non-methane hydrocarbon values exceeded the full scale measurement capacity of the recording system. The pattern shown, with an initial rise in levels between 9.00 p.m. and midnight, and a decreasing trend towards dawn on the following day, is typical of these events.

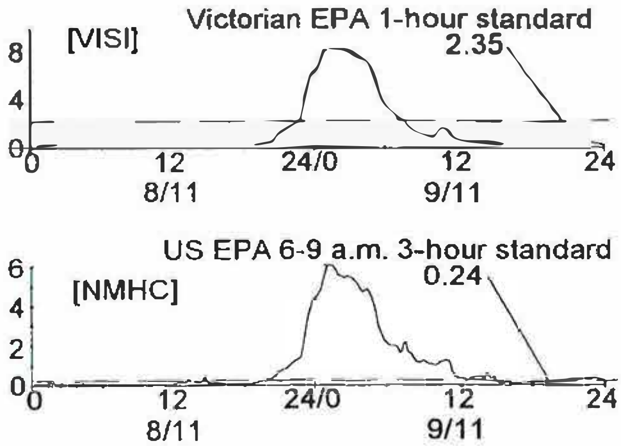

Figure 2. Visibility ('[VISI]', units of 10^{-4} m^{-1}) and non-methane hydrocarbon ('[NMHC]', parts per million) at Hope Valley, during 8-9 November 1990. The horizontal axes represent two days of time, from midnight to midnight, and hours through the day.

In previous summers of recording, smoke levels have cleared before the afternoon ozone peak might have developed. But during the spring and summer of 1992-93, two ozone events occurred in association with bushfire smoke. Figure 3 presents time sequences of visibility and ozone measurements, obtained at the DEP's Caversham site, for the first event, on 22 October 1992.

In this case, and the other on 13 January 1993, the visibility standard was approached, but not exceeded during the period of increased ozone concentration. Also in both cases, the range of standards currently recommended for ozone was exceeded.

The DEP is monitoring ozone and nitrogen oxide levels across the metropolitan area, as a major component of a study funded by the State Energy Commission of WA. At the time of the October 1992 event, equipment was operational at Caversham, Swanbourne, Kenwick and Quinn's Rocks (See Fig. 1 for locations). The peak hourly averages measured were 107, 106, 86 and 74 ppb respectively, showing that increased photochemical smog levels occurred across the whole metropolitan region, both upwind and downwind of the urbanized area. (Winds at the time were from the south-west.)

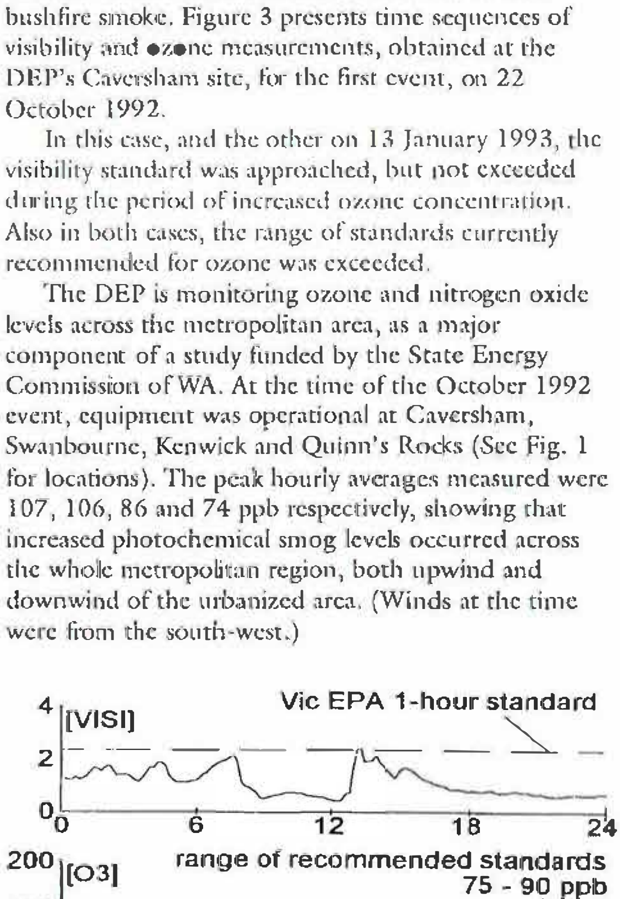

Figure 3. Visibility ('[VISI]', units of 10^{-4} m^{-1}) and ozone ('[O3]', parts per billion) at Caversham, during 22 October 1992. The horizontal axis unit is hours through the day.

While ozone levels did not vary greatly across the metropolitan area in the October episode, the January event showed large variations. Peak levels measured the DEP's sites were as follows:

Rottnest Island	28	ppb at	11.30 a.m.
Swanbourne	36		11.40
Quinn's Rocks	35		11.50
Two Rocks	23		11.50
Cullacabardee	85		12.10 p.m.
Kenwick	62		12.20
Caversham	85		12.10
Rolling Green	112		2.20

Highest values occurred inland, and to the north of the city. Winds at noon were 7-8 m s^{-1} from about 210° at the coast, and 4-5 m s^{-1} from 230-240° inland. At these speeds, the time of travel of an air mass from Swanbourne to Cullacabardee (about 20 km downwind) would have been close to an hour. During this time the ozone concentration in the air mass rose by about 50 ppb. This represents a high rate of reaction, compared with normal Perth experience.

The differences between the two events may be linked to the differing meteorology of the two days. On 22 October, there was an initial light easterly wind at the coast, with a light southerly at Rottnest. The southerly wind was due to the presence of a common feature of Perth's summer climate, the 'west coast trough', a low-pressure trough that forms offshore with its axis parallel to the coast. The occurrence of southerly winds at Rottnest shows that its axis lay between there and the coast.

This pattern, during summer, is the most common one leading to a photochemical smog episode. Morning urban emissions of VOC and nitrogen oxides are carried offshore by the easterly, but held close to the coastline by the effects of the coastal trough. Reactions within the offshore air mass permit increased ozone concentrations to form, before the air is returned onshore by the sea breeze. Levels of nitrogen oxides measured at Swanbourne on 22 October were consistent with values preceding such an event.

It therefore seems likely that the increased ozone levels arose from the mixing of normal urban emissions with VOC in the offshore smoke. The probable high reactivity of the mix would have compensated for the lower reaction rates in spring, resulting in ozone concentrations more typical of summer events.

By contrast, the 13 January winds were initially light southerly, with an onshore trend developing during the morning. These conditions do not normally lead to increased ozone levels. A steady southerly would carry morning emissions well out of the metropolitan area before increased ozone concentrations could form.

The moderate levels of ozone and low concentrations of nitrogen oxides measured at the coast on 13 January were consistent with these conditions. The development of high ozone concentrations inland could only be ascribed to much higher reactivity of VOC in the smoke.

ILLUSTRATION OF SMOKE TRAJECTORIES

To the best of the DEP's knowledge, the October and January events originated from different sources. The sole major burn conducted before the first event was on private land, north-east of the city. Other minor burning-off operations were also in progress in the northern suburbs. Before the second event, the Department of Conservation and Land Management had conducted a significant controlled burn in the far south-west of the state.

It was believed that in both cases, a major factor in common was the presence offshore of a coastal low-pressure trough. Circulation of air southward to the east of the trough, and northward to its west, creates the potential to trap smoke for an extended period. This trapping may be enhanced when a closed low forms in the trough, as happened at least for the October event.

To illustrate the potential effects of the trough, a three-dimensional mesoscale model was run from an initial easterly-wind condition, typical of the start of a period of trough development. The model used was an extension of a two-dimensional one developed for both forecasting and analytical purposes (Rye 1989) which had been extensively validated in the Perth region, and has recently been applied for sea breeze and trough studies related to the current Perth Photochemical Smog Study. In the most representative case so far, the trough formed and deepened over a period of three days, the fourth and fifth day of model time being used in the work quoted here.

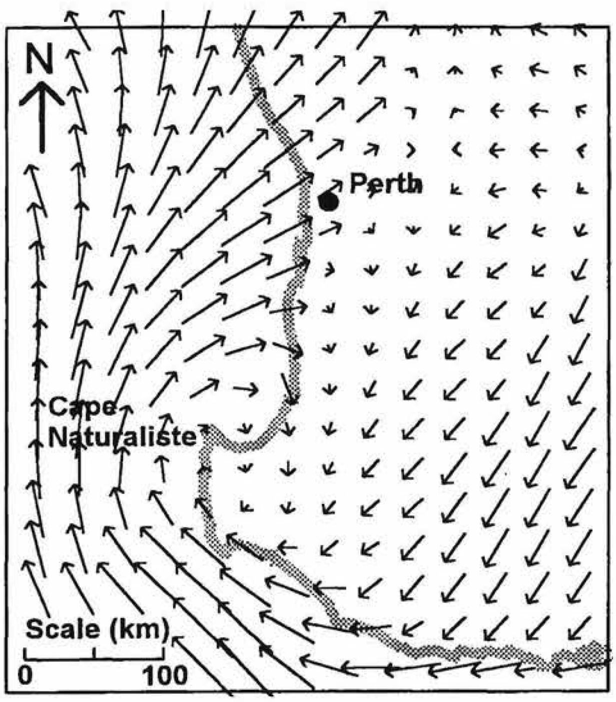

Figure 4. Wind vectors for 9.00 a.m. on the fifth day of model time, showing circulating wind flow.

Figure 4 shows a wind vector plot for a subsection of the modelled region at 9.00 a.m. on the fifth day, as the trough was passing inland. The closed circulation near Cape Naturaliste is a common component of the trough.

The calculated hourly meteorological fields, from which Figure 4 is an extract, were used to estimate the movement of smoke that might have been released from a location in the south-west forest region.

In the modelled case the trough passed inland slightly later than the time that would have brought the smoke trajectories to Perth. Therefore, to illustrate the type of event that was observed on the 13 January, a slight shift of smoke emission time was required.

Normally, controlled burns are conducted during the daytime, but in this exercise it was necessary to shift the period to cover the times 12.00 noon to 8.00 p.m.

The complications evident in the trajectories shown in Figure 5 are probably typical. A particle emitted at 7.00 p.m. (hour 19) would have been carried northward in the evening for the fourth model day, south-west then north-west overnight, then north-eastward through the next day. By contrast, the trajectory which commenced an hour earlier moved initially south-west, then north-west overnight, and finally to the north-east.

The implication of this analysis is that there is potential for smoke to reach Perth not only from close sources, but also from large distances away. However, for photochemical smog to form when the source is distant, a low pressure trough must be present offshore. In these conditions, accurate trajectory forecasts are unlikely.

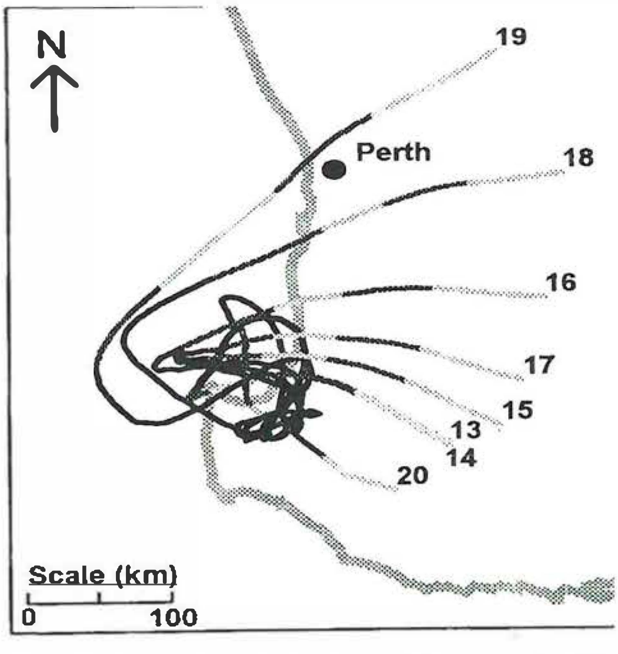

Figure 5. Trajectories calculated up to 6.00 p.m. on the fifth modelled day, for particles released in the south-west. Each trajectory is labelled with the hour of the previous day of its release. Contrasting shades show the final four three-hour periods on each.

DISCUSSION

The additional effect of bushfire smoke, in contributing to episodes of photochemical smog, introduces a complication to the management of both smoke and smog on the Perth metropolitan area. There are a number of factors to consider, including the following:

Seasonal Weather Factors For normal sea breeze strengths and directions, smoke from south-western sources passes the Perth region overnight. To pass the Perth metropolitan area during the day time, the path of the smoke must be extended by some form of recirculation, or the winds carrying the smoke must be significantly lighter. Both of these conditions normally require the presence of a low pressure trough along the west coast.

During summer, when such a trough is present, temperatures on the coast are generally above 35°C, higher than those considered by the Department of Conservation and Land Management to be ideal for controlled burns. In addition, private burning is prohibited through most of summer. For these reasons, such events should be rare in the height of summer.

But in the early spring period, conditions suitable for controlled burns can arise when temperatures are higher than normal. In these cases, careful consideration of possible smoke trajectories will be essential to minimize the chance of a photochemical smog event in Perth.

However, there is currently no formal requirement that private land-clearing burns be conducted with the same attention to weather constraints as those of the Department of Conservation and Land Management. The 22 October event occurred shortly before the commencement of summer-season fire restrictions. For the recurrence of such an event to be prevented, extra controls on the lighting of large fires may be required when a deep low pressure trough develops off the west coast.

Trough Occurrence and Forecasting The potential for events described here to produce photochemical smog requires the presence within the trough of a recirculating flow. While many coastal troughs form through a summer, most pass inland without forming such a flow pattern. Others cross the coast at a time of day when smog formation is prevented or limited by the lack of sunlight. The Bureau of Meteorology has long regarded the accurate prediction of movement of the trough as a high priority, and the potential effect of a trough passage in the crucial time range from about 6.00 a.m. to 3.00 p.m. provides a further reason for priority treatment.

Given the high variability of smoke trajectories, an accurate prediction of movement of smoke over Perth due to the passage of a trough is unlikely. Any control strategy based on weather predictions would therefore require inclusion of a wide margin of safety.

Future Trends Current vehicle emission controls are tighter on hydrocarbon than on nitrogen oxide emissions, so a long-term reduction in emissions of hydrocarbons from the vehicle fleet can be expected, but less so of nitrogen oxides.

The cause of the events described here was a large input of hydrocarbons to the urban atmosphere. Under conditions of surplus hydrocarbons, the principal controlling factor is the supply of nitrogen oxides. Given the lesser reduction of nitrogen oxide emissions, there appears little prospect of a reduction over time of the frequency of smoke-originated smog events.

CONCLUSIONS

The two events described here were the first major episodes of photochemical smog detected in three years of monitoring, to which a bushfire contribution could be ascribed. Nevertheless, both occurred due to a combination of smoke with a coastal low pressure trough, the occurrence of the latter being a normal part of Perth's summer weather pattern.

Measurements suggest that the smog developed due to a mixture of urban nitrogen oxide emissions with reactive organic compounds in the smoke. In one case, the mixing occurred offshore, in the other it occurred as the smoke passed over the Perth region.

Given the potential effect of the combination of bushfire smoke and the coastal trough on urban air quality, and the very specific circumstances involved, it should be possible to include its consideration within the smoke management program already operated by the Department of Conservation and Land Management. But additional controls may be required to ensure that events such as occurred in October, arising from smoke from private burning-off operations, do not recur.

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Smoke management in Western Australia's south-west forests

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ABSTRACT

Prescribed burns conducted by the Department of Conservation and Land Management (CALM) on forest blocks up to 300 km south of Perth, can result in the development of smoke haze in the Perth metropolitan area. This smoke haze can significantly reduce visibility along highways and airports, and upset residents who perceive there may be health risks associated with smoke concentration.

Active smoke management programs for burns close to Perth began in 1974, and smoke management has since been extended to all forest areas in the south-west of the State.

Studies on the impacts of various weather parameters and operational factors on the incidence of smoke haze in Perth indicated that the most important factors included atmospheric stability conditions, wind directions, location of burn in relation to other burns, accumulated areas of burns, timing of burns, and distance of burn from Perth.

A set of smoke management guidelines have been developed and tested successfully over three fire seasons. The guidelines have been incorporated into a decision model for fire managers. The model is presented in the form of a series of simple decision charts. Further research into prediction of smoke transport and dispersion is being undertaken between CALM and the WA Regional Bureau of Meteorology.

INTRODUCTION

The Department of CALM carries out an extensive prescribed burning program in the south-west forests, in order to reduce flammable fuels and mitigate the undesirable social, economic, environmental and human problems caused by destructive wildfires.

In part, CALM's burning program stems from the direction to do so, made by a Royal Commission (Rodger 1961) which enquired into the widespread bushfires in summer 1961, which destroyed Dwellingup and several other settlements.

In part, CALM is also responding to the Bush Fires Act of 1954 which requires landowners to take responsible action to minimize fire hazard. While CALM is not a landowner in this sense, it has a duty of care as a neighbour to carry out appropriate works which reduce the probability of fire spreading onto neighbouring properties.

Prescribed burning is carried out by CALM District staff, often with the assistance of volunteer bush fire brigades. The work is done in accordance with a Fire Management Plan for the district which sets out the areas to be burnt and the frequency and season of the burns. Fuel reduction burns are concentrated in the areas where the highest values need to be protected from potential wildfires. Other planned burns are lit for a variety of purposes, including wildlife habitat management and forest regeneration (McCaw and Burrows 1989).

The fuel-reduced areas which result from prescribed burning have enabled fire fighters to suppress many potentially serious wildfires, to the extent that there have been very few major fires in the past thirty-three years in the jarrah and karri forests of Western Australia (Underwood *et al.* 1985).

During this time no fire fighter has been burnt to death in a forest fire in Western Australia, as has occurred in south-east Australian states, nor has there been any civilian loss of life.

Land managers have a responsibility to ensure that prescribed burning is strategic, timely, appropriately and correctly applied. An additional responsibility of land managers extends to the minimization of impacts of smoke from prescribed burning on the community. Smoke from these burns can be more than a nuisance as it can reduce visibility along highways, close airports and upset metropolitan residents because of perceived health risks associated with smoke concentration.

SMOKE AND WEATHER CONDITIONS

Fuel reduction burns in the south-west forests of

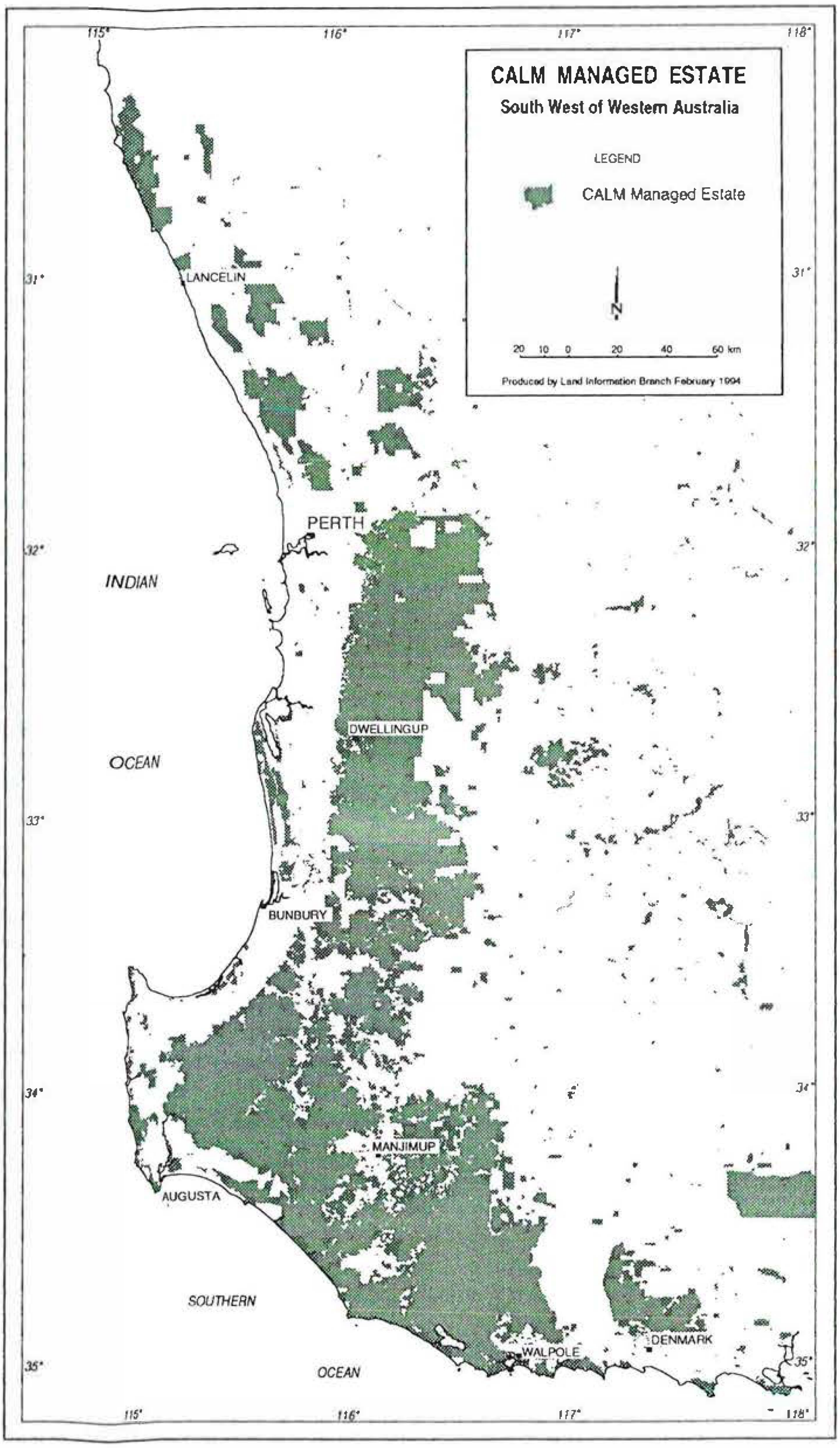

Figure 1. Areas of CALM-Managed Estate in south-west of Western Australia.

Western Australia are normally carried out in the spring, early summer and autumn months early in the cycle of anticyclones which move from west to east across the southern half of the continent. During these periods moderately warm and dry conditions occur for periods of 3 to 5 days enabling low intensity burns to be carried out safely. The range of conditions typically associated with prescribed burning for fuel reduction are shown in Table 1. The typical range of fire behaviour parameters observed at these prescribed burns is shown in Table 2.

Burns in the northern jarrah forest north of Bunbury can be carried out on the westerlies which occur as the anticyclone begins its easterly movement across the bottom part of the State. The south-easterly winds are most often used for the forest zones south of Bunbury as these coincide with moderate weather conditions and suitable levels of fuel moisture.

The winds in the southern quarter provide the safest conditions under which to light prescribed fire as these conditions are generally characterized by mild temperatures, relatively moist fuels, predictable and moderate wind speeds and relatively stable atmospheric conditions. Unfortunately those conditions that are normally suitable for a safe effective fuel-reduction burn in the forest are most often the same that lead to poor smoke dispersal and the accumulation of high levels of smoke concentration.

South-easterly winds transport smoke from burns in the northern jarrah forests directly into Perth. Further south, smoke can be carried by the south-easterlies over the Indian Ocean from where some of the smoke can be blown back over the coast and Perth by the late afternoon south-westerly sea breeze.

During days of very stable atmospheric conditions this has resulted in a smoke haze in Perth when the visibility index exceeds the current standard applied by the Department of Environmental Protection (DEP). This standard is the visual range of 20 km averaged over 1 hour. This standard is determined by reference to the light-scattering properties of fine airborne particles as measured by a nephalometer. A visibility coefficient is determined from the nephalometer readings. A coefficient value of 2.3 is equivalent to a visual range of 20 km. Values which exceed 2.3 when averaged over 1 hour are considered to exceed the air quality guidelines adopted by the DEP. The smoke haze usually lasts for 2 to 8 hours overnight and disappears in the morning with the dissipation of the nocturnal inversion and the onset of the prevailing easterlies.

High intensity wildfires normally occur on north-east, north or north-west winds. While such fires generate much larger volumes of smoke that can affect southern regional centres such as Bunbury, Manjimup and Denmark, the smoke is normally carried south away from the Perth metropolitan area. Any 'pollution problem' associated with bush fire smoke in Perth is therefore more related to prescribed burning than to wildfires.

MANAGING THE SMOKE

CALM has managed smoke in the northern jarrah forest, within 100 km of Perth, for nearly twenty years. Based on smoke distribution studies conducted by the Commonwealth Scientific Industrial Research

TABLE 1

Range of weather and fuel conditions used for fuel reduction burning operations in jarrah and karri forests

	TEMPERATURE (°C)	RELATIVE HUMIDITY (%)	WIND DIRECTION	WIND SPEED (km h^{-1})	SURFACE FUEL MOISTURE CONTENT
Range of Condition	20 to 30	30 to 50	W-S-E	10 to 20	10 to 16%

* Moisture Content expressed as percentage of oven-dry weight

TABLE 2

Range of fire behaviour characteristics for fuel reduction prescribed burns in forest fuels of WA

FIRE BEHAVIOUR CHARACTERISTICS	FLAME HEIGHT (m)	HEADFIRE RATE OF FORWARD SPREAD (m h^{-1})	FIRELINE INTENSITY (kW m^{-1})
Range of Parameters	0.3 to 2.0	15 to 40	100 to 500

Organisation (CSIRO) in conjunction with the Forests Department of Western Australia in the early 1970s (Vines *et al.* 1971), a predictive model was developed to make reliable predictions about the fate of smoke plumes from planned burns. The model has been used to determine the maximum size of an area that can be burned under various wind directions at a predetermined distance from Perth so that the air quality in the metropolitan area is not significantly affected. For example, under south-easterly and easterly winds, areas burnt within the northern jarrah forest zone are restricted to a size given by the following simplified equation.

Max. Burn area (ha) = Distance from Perth Airport (km) x 20.

For example, the largest burn area for a location 80 km south-east of Perth Airport is 1600 ha.

This decision rule is applied particularly where winds are likely to lead smoke directly into Perth. Larger burns can be conducted only when winds are predominantly from the south-west. Stringent application of this model over the past twenty years has resulted in very rare instances when smoke from northern forest burns has blown directly onto Perth, usually as a result of an incorrect wind forecast.

In 1990 CALM became aware that smoke haze from southern forest burn operations was exceeding visibility standards on seven or eight occasions each summer. These instances, which occurred mostly at night, were detected on three DEP nephalometer recorders located in and around Perth. The origin of these smoke sources were confirmed by LANDSAT images to be as far as 300 km south of Perth.

As a result, CALM has undertaken studies into determining the relationship between weather and burn operation factors and the incidence of smoke haze from these southern forest burns (Smith 1991, 1992).

These studies identified a number of factors that may contribute to the incidence and extent of smoke haze in the Perth metropolitan area.

(i) Atmospheric Stability

Atmospheric stability as measured by the rate of temperature change with height. The presence and depth of low-level temperature inversion measured at 7.00 a.m. each day at Perth airport provided a good guide to the probability of smoke haze occurring in the metropolitan area. The stronger the inversion temperature change, the more likelihood that smoke will accumulate and exceed limits.

(ii) Total Area Burnt Within a Day

The area of prescribed burning that can be undertaken without resulting in smoke haze appears to be dependent on the atmospheric instability. Under weak inversion (<3°C change within the first 1000 m) the accummulated area of several burns can exceed 15 000 ha. This area is reduced to about 12 000 ha under moderate inversion (<3-6°C change); and 6000 ha for strong inversion (7-10°C change). Inversion temperature changes in excess of 10°C were rare and often associated with a west coast trough that prevented any dispersion of smoke haze on the west coast.

(iii) Burn Concentration

The concentration of smoke was related to the geographic distribution of burning operations. Separation of burns by more than 80 km reduces the smoke concentration and therefore the likelihood of high smoke haze levels recorded in Perth.

(iv) Wind Direction

The transport of smoke from southern burns into Perth occurs predominantly under southerly winds. Afternoon seabreezes have substantial influence on the occurrence and timing of arrival of smoke haze in Perth.

Winds with a northerly component reduced the likelihood of smoke haze in Perth, but can disperse smoke over regional centres in the lower south-west.

(v) Scheduling of Burns

Delaying the ignition of burns can reduce the chances of smoke reaching the Perth area. In some circumstances, smoke from burns ignited later in the day is blown over the Indian Ocean and is not widely dispersed because the influence of the seabreeze has diminished by the time the smoke reaches Perth.

SMOKE MANAGEMENT GUIDELINES

Smoke management guidelines have been developed over the past three years which take into account atmospheric conditions, wind direction, forecast of seabreeze, location of burn in relation to metropolitan area, the size of burns, location in relation to other large CALM burns, total area likely to be burnt on the day, burn priority and recent history (e.g. already partly alight), availability of resources including aircraft for ignition, and incidences of other burn operations undertaken by other agencies and landholders.

To assist fire managers co-ordinate the complex burning program in accordance with these smoke management guidelines, a decision model has been recently developed and applied in 1992-93 and will be further tested in the 1993-94 burning season. This model is presented in the form of a series of simple decision charts (Figs 2 to 7).

The application of the smoke management guidelines during the 1992 and 1993 burning seasons resulted in a low incidence of smoke haze in Perth. There was one serious haze occurrence that plagued Perth for three days from 21 to 23 October 1992 (Rye 1994). This appeared to be a result of an unusually intense inversion over the Swan Coastal Plain which trapped smoke from wildfires and private burning operations to the north of Perth. CALM's burning operations were cancelled during this period to avoid contributing to the problem.

PRESCRIBED BURNING AND SMOKE MANAGEMENT DECISION SYSTEM FOR SOUTH-WEST FORESTS OF WA

POSSIBLE PATHWAYS OF SYSTEM

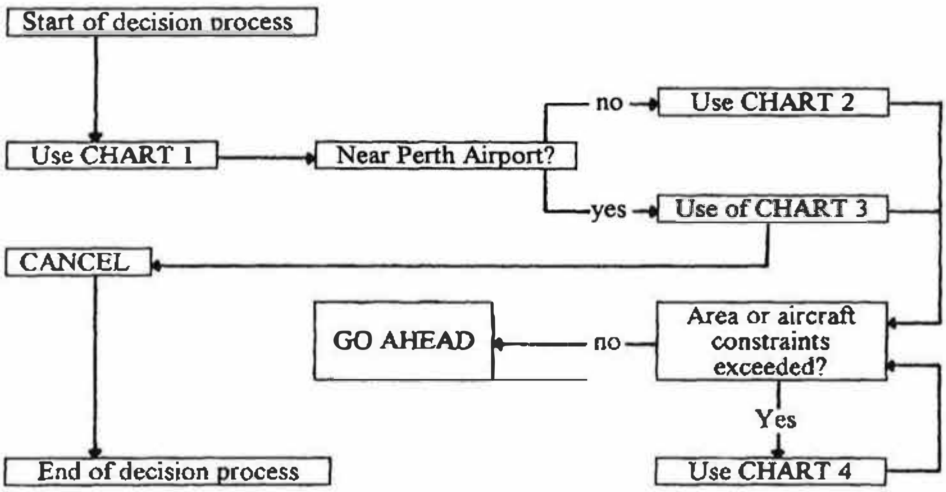

Figure 2. Decision Chart for Smoke Management.

PRESCRIBED BURNING AND SMOKE MANAGEMENT - DECISION CHART 1

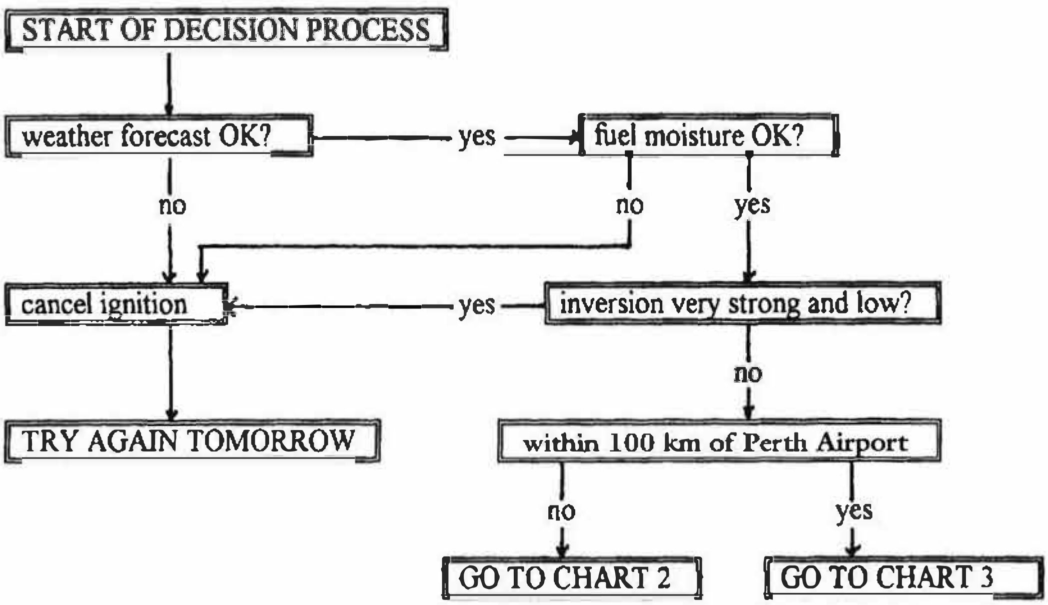

Figure 3. Decision Chart 1.

PRESCRIBED BURNING AND SMOKE MANAGEMENT - DECISION CHART 2

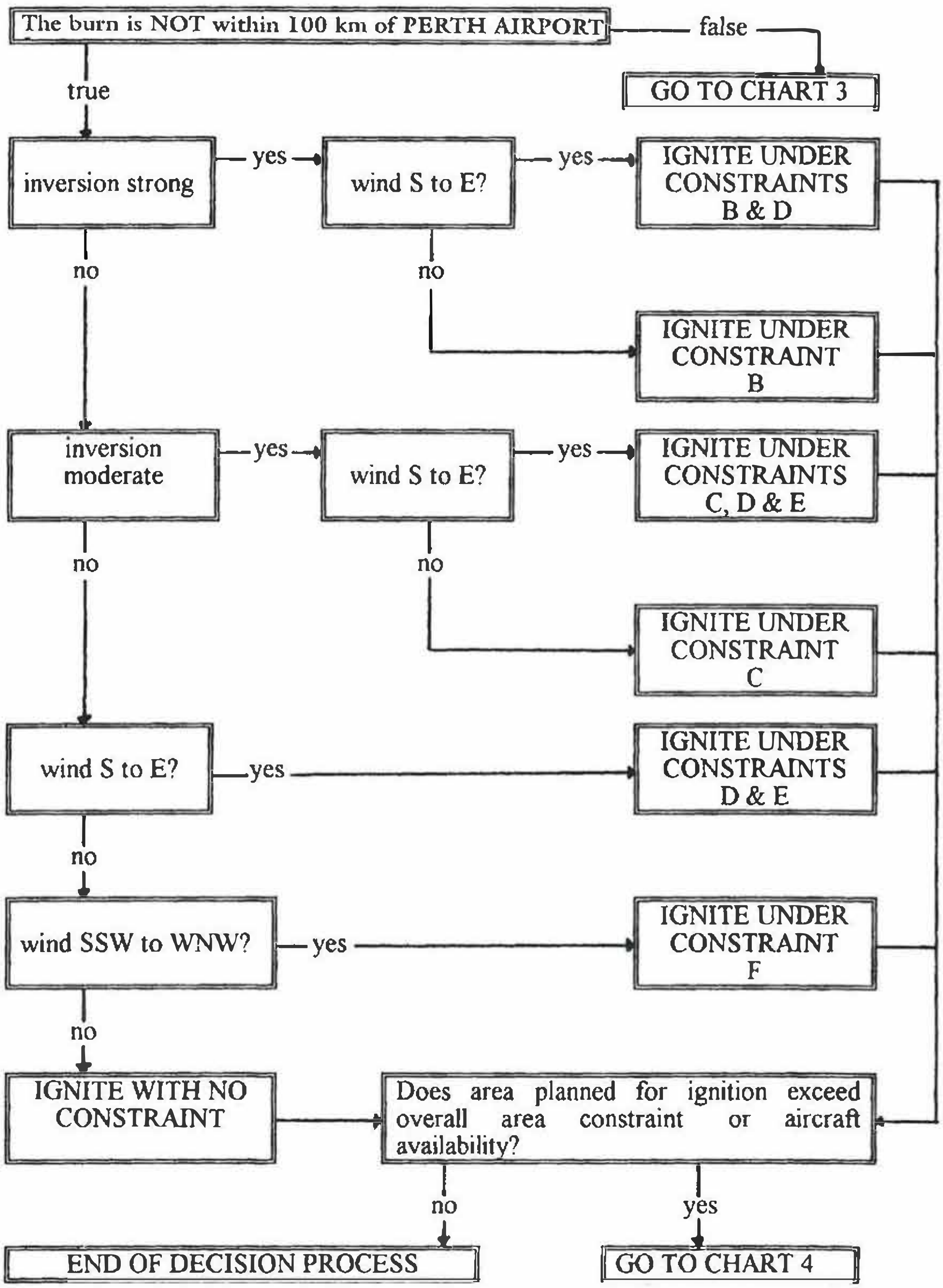

Figure 4. Decision Chart 2.

PRESCRIBED BURNING AND SMOKE MANAGEMENT - DECISION CHART 3

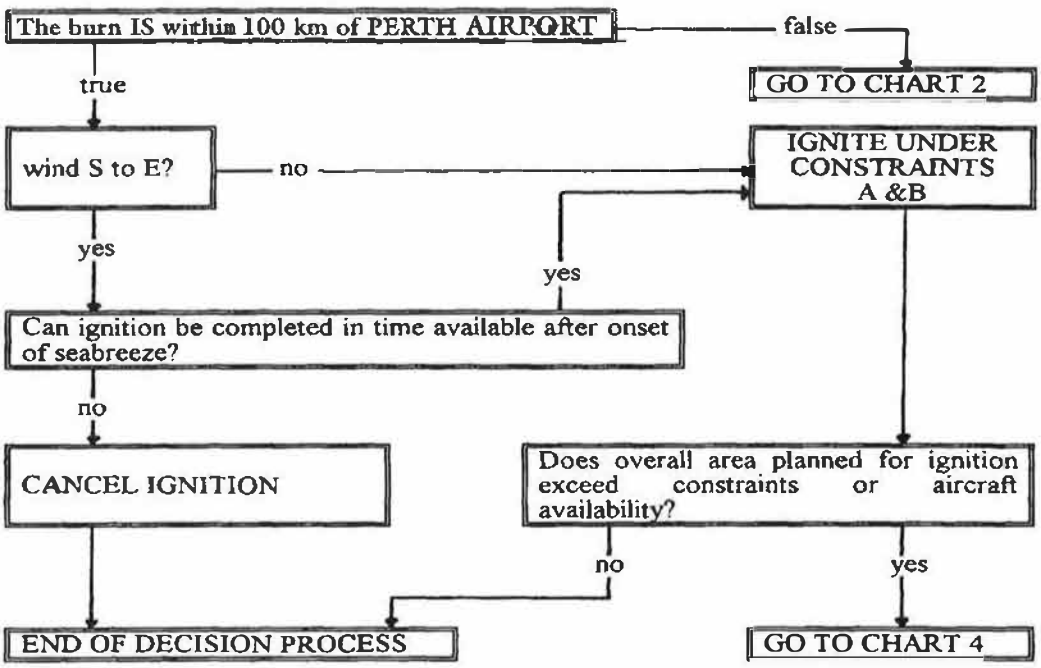

Figure 5. Decision Chart 3.

PRESCRIBED BURNING AND SMOKE MANAGEMENT - DECISION CHART 4

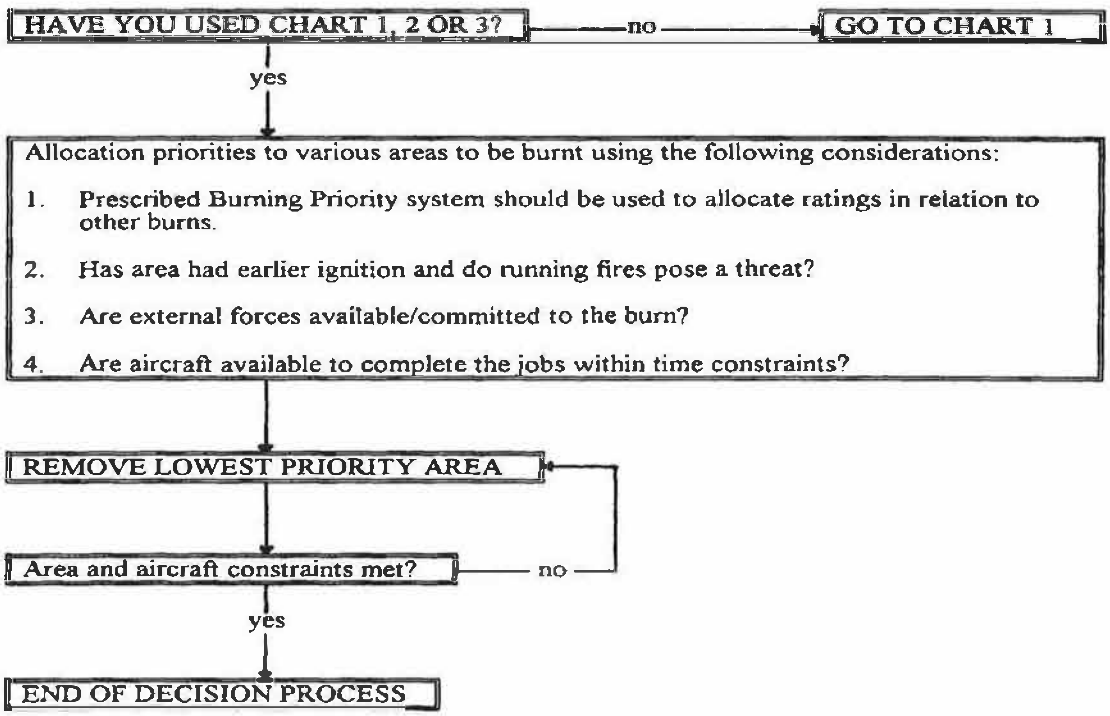

Figure 6. Decision Chart 4.

CONDITIONS OF BURNING

A....Individual burn areas limited by ratio rule*.

B....Total area burnt not to exceed 6,000 ha.

C....Total area burnt not to exceed 12,000 ha.

D....Burns greater than 4,000 ha to be at least 40km from any other burn on first and following day.

E....Total area burnt within any 80km radius not to exceed 15,000 ha.

F....Total area burnt within any 80km radius not to exceed 15,000 ha

* Area = 20 x Distance to Perth Airport (km)

Figure 7. Limitations on Burning for Smoke Management.

FUTURE DEVELOPMENTS IN SMOKE MANAGEMENT

Monitoring of smoke haze incidence in Perth during the 1992 and 1993 burning seasons indicated that the application of simple decision rules has significantly improved smoke management from prescribed burning in Western Australian forests. However, more remains to be done. Unpredicted weather events can still surprise weather forecasters and fire managers, and re-ignition of unburnt fuels on subsequent days can cause smoke to accumulate over population centres.

There is still much to learn about smoke transport and dispersion. CALM and the Bureau of Meteorology are to conduct a joint study that aims to provide a means of accurately predicting the trajectories of smoke parcels on the days of proposed burns. The Bureau's current operational numerical model (RASP) runs a trajectory plotting routine, but the model's grid resolution (150 km) is too coarse to capture the detail required. It is proposed a mesoscale numerical model with a grid resolution around 10 km be developed by the Bureau to predict the transport of smoke emitted from sources throughout the south-west.

In addition, the Bureau is investigating the synoptic situations during which smoke haze problems have occurred. It is hoped this will help improve the identification of dispersion factors and the development of a Dispersion Index similar to that used in South-eastern United States (Lavdas 1986).

CONCLUSION

While the applications of smoke management guidelines and appropriate burning prescriptions will help to minimize the occurrence of smoke haze over Perth, there is no guarantee that it can be eliminated altogether.

The complete cessation of prescribed burning may have short-term impact on the incidence of smoke haze, but will inevitably create another social problem; the cost and tragic impact of destructive wildfires on communities, properties and forest ecosystems in the south-west. It is simply not acceptable to put rural communities and our natural assets at risk to wildfire damage and destruction in an attempt to eliminate occasional bush fire smoke from the city.

ACKNOWLEDGEMENTS

The prescribed burning and smoke management decision system was developed by David Ward, Science and Information Division of the Department of Conservation and Land Management, Como, Western Australia.

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Responses of plant populations to fire: fire season as an under-studied element of fire regime

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ABSTRACT

It is well known now that the responses of plant populations to fire depend on the fire regime. Different fire intensities and fire frequencies can both alter plant population dynamics, but other aspects of the fire regime have received less attention. We have set up a replicated study to examine how season of burning might affect plant populations by altering recruitment in Hawkesbury Sandstone vegetation. Time to germination and germination rates of sown seeds of *Banksia* and *Hakea* were measured after fires in two different springs and two different autumns. Recruitment varied markedly between fires in the same season but different years. Unlike studies in Mediterranean-climate regions, the timing of germination after fire was not predictable. We conclude that the vagaries of post-fire climate introduce a stochastic element into the plant population response to fire that should be incorporated into population models and fire management strategies.

INTRODUCTION

Many characteristics of plant species in fire-prone ecosystems are considered to have evolved in response to an historic fire regime - with fires of an intensity, frequency and season that are typical of a given region (Gill 1975). Gill's work (e.g. Gill 1975, 1981, 1989) has emphasized the importance of focussing on fire regime in interpreting and predicting the ecological impacts of fire.

Considerable experimental and modelling work has been done on the effects of several components of fire regime on plant populations and communities. This work has focussed on the effects of varying fire intensities and fire frequencies on mortality of plants and recruitment to plant populations (e.g. Bradstock and Myerscough 1981; Bradstock and O'Connell 1988; Auld 1986), perhaps because these aspects of fire regime are the most conspicuous ways in which fire regimes prescribed for some land-management objectives differ from natural fire regimes.

Prescribed fires also differ in the season in which they are applied, the cool-season being favoured because deliberate fires can be more readily controlled. In this paper, we explore the possible ecological effects of season of burning by presenting some preliminary data from a pilot study designed to examine time to germination and magnitude of germination in two Proteaceae species in the Wollongong area.

Empirical studies of the effects of fire season, mostly conducted in Mediterranean-climate ecosystems with predictable timing of wildfires and strong climatic seasonality, suggest that seedling densities are lower after fires in winter/spring than after summer/autumn fires (e.g. McMahon 1984; Bond *et al.* 1984; Midgley 1989). One mechanism that has been proposed to explain this pattern is that the post dispersal seed losses are greater after a spring fire (see Bond 1984; Cowling and Lamont 1987). It is argued here (i) that germination occurs because of the favourable conditions in winter/spring, and (ii) that seeds are at risk while they lie on the soil surface after dispersal and prior to germination. Hence seed losses would be minimized after autumn fires, because seed release is immediately followed by germination. After spring fires, seeds must see out summer and autumn before germination.

As Bradstock and Bedward (1992) pointed out, much of the information on the effects of season of fire (with some exceptions – see Bond *et al.* 1984) comes from single studies of fires that are unreplicated within years or seasons. Moreover, there is no reason to expect that the processes determining the patterns of recruitment are similar in other ecosystems.

In this study, we conducted an experiment with replicated fires in each of two springs (1990 and 1993) and two autumns (1992 and 1993). We measured the time from seed release until germination and also the proportion of seeds germinating after each fire.

METHODS

In 1990, an 8100 m^2 site of Hawkesbury Sandstone vegetation was selected near Picton, New South Wales. This woodland with a heath understorey is typical of many sites in the Sydney region and contained extensive populations of two woody shrub species that were the focus of the study: *Hakea sericea* (Harden 1992) and *Banksia spinulosa* (George 1984). The site was divided into 36 plots, each between 0.1 and 0.2 ha. This patch area is typical of hazard-reduction burning carried out by the Balmoral Bush Fire Brigade in the general area of the study (I. Tait, personal observation). The season-of-fire treatments (initially to be one each of spring, autumn, summer) were allocated randomly to plots. After the first spring (1990) and autumn (1992) fires, it was decided that replication of these seasons of burning was important, so the third group of 12 plots was divided into two sets of 6, randomly allocated to either spring 1993 or autumn 1993 burning.

Fires in the replicate plots for each year/season were held within a few days of each other, and all were burned in the mid to late afternoon in a manner typical of a routine hazard reduction fire. Fire intensity varied both between and within plots, with canopy scorch in some areas and only patchy burning of leaf litter in others. Seeds released from fruits heated over a fire outside the site were sown on the ground in each plot. In each plot, 20 seeds of each species were dropped from a height of about 30 cm into a 50 x 100 cm grid, and their precise locations in the grid recorded. Sowings were completed 2-3 weeks after the fires, which is when natural seedfall was occurring. The grids were visited regularly to record the appearance and survival of seedlings.

A major rainfall event following the spring 1990 fire re-assorted seeds, litter and soil, such that continuing germination could potentially be confused with seed rain from surrounding plants. Seedlings from sowings in the other fires remained in their locations and were readily identified in consecutive censuses. The surrounds of each seedling grid were searched thoroughly for seedlings, confirming that 'background' germination levels were negligible.

RESULTS

Following the fires in spring 1990, the time to first germination was 10 months after sowing for both *Hakea* and *Banksia* (Fig. 1a). Thus seedlings first appeared in August 1991. Over 25 per cent of *Hakea* and 10 per cent of *Banksia* seeds germinated, but survival over the ensuing 1.5 years was low, with just over 5 per cent of the original seeds still remaining as seedlings.

Germination following the fires in autumn 1992 did not occur, as expected, in the following winter (Fig. 1b). Seedlings of both species first appeared 7 months after sowing: in January 1992. New seedings continued to appear for a further 7 months with very little mortality. In contrast to the spring 1990 fire, a very low proportion of seeds germinated: less than 5 per cent for *Hakea* and less than 2 per cent for *Banksia*.

After the autumn 1993 fires, germination had already started at 4 months: i.e. in September 1993 (Fig. 1c), though it is not yet possible to assess the magnitude of germination. The spring 1993 fires have just occurred.

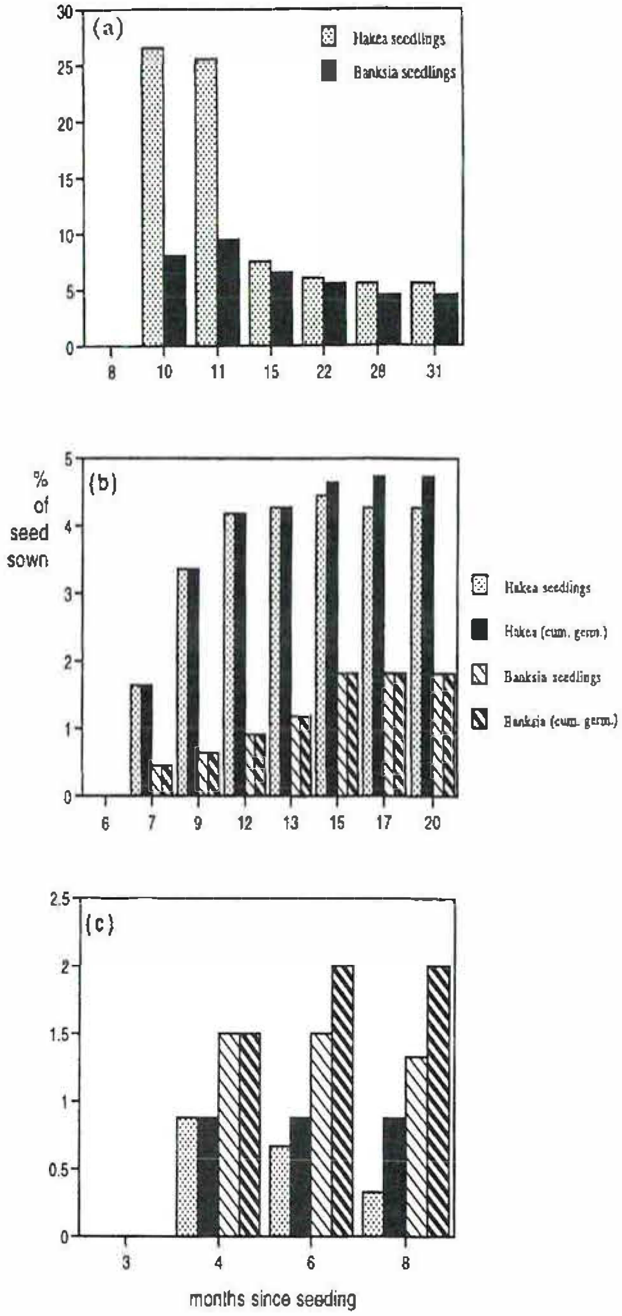

Figure 1. (a) Changes in numbers of Hakea sericea *and* Banksia spinulosa *seedlings with time after fires in spring 1990; (b) and (c) cumulative germination and numbers of seedlings alive with time after fires in autumn 1992 (b) and autumn 1993 (c).*

DISCUSSION

The timing of first germination after the 1990 spring fires was indeed delayed until the following winter. This is the pattern that would be expected from Bond (1984) and Cowling and Lamont (1987). The long-term results of the 1993 spring fire remain to be seen, but there had clearly not been substantial germination by summer 1993-94. The time to germination differed between the two autumn fires. There was a delay of over 7 months before germination occurred after the autumn 1992 fires. Thus seedlings started to appear in summer. In contrast, germination had started just 4 months following the 1993 autumn fires. Even with these three sets of fires, it is apparent that the timing of germination is not strongly seasonal. It is likely that the less predictable season bringing substantial rainfall in the Sydney region, in contrast to Mediterranean-climate regions, is the major factor controlling the season of germination. The coincidence of germination with the first period of substantial rainfall after the first two sets of fires (spring 1990, autumn 1992) lends support to this hypothesis (Fig. 2).

Although the time to germination was greatest after the spring 1990 fires, the proportion of seeds germinating was not substantially greater than after the other fires (Fig. 1a; nearly 30 per cent of seeds for *Hakea* and 10 per cent for *Banksia*). Poor germination was expected to result from the long period during which seeds were at risk from predation, fungal attack and other hazards. Poor germination did occur following the 1993 autumn fire (less than 5 per cent for *Hakea* and less than 2 per cent for *Banksia*), where there was also a substantial delay between the fire and germination. It is not yet possible to assess the timing or magnitude of germination following the autumn 1993 fires.

The main conclusion of these preliminary studies is that germination is not strongly tied to the winter/spring season. Nor does there appear to be a strong inverse relationship between the proportion of seeds germinating and the length of time between a fire and germination. In this environment, therefore, the pattern of post-fire rainfall, irrespective of season, appears the most likely determinant of the timing of germination. The factors determining the magnitude of germination are still unclear. One possibility is suggested by the observation of reassortment of seeds and litter into soil depressions, following the major rains after the 1990 fires. This process may have deposited seeds in microsites that were especially suitable for germination (see Whelan 1986; Enright and Lamont 1989; Lamont *et al.* 1991; Battaglia and Reid 1993).

The building of realistic models of population dynamics under various fire regimes will clearly require careful inclusion of the impact of factors such as post-fire rainfall (Bradstock and Bedward 1992), and other factors that might affect timing and amount of germination, such as spatial heterogeneity of safe sites, and post-dispersal seed removal.

CHALLENGES FOR MANAGERS

Understanding the effects of different fire seasons on plant demography and then incorporating this information in management offer enormous challenges for land managers, especially in situations where conservation of plant species and communities is a significant management objective (see Whelan and Muston 1991).

If the timing of prescription burning does influence recruitment to populations of some plant species, then wide-spread hazard reduction burning, especially in winter/spring, may be expected to have detrimental effects. However, the preliminary results of our study indicate that reduced recruitment after spring fires may not be a general effect. The differences between this study and previous work in Mediterranean-climate systems reinforce the view (Williams *et al.* 1994) that management *prescriptions* are not likely to be 'portable', depending on the characteristics of a particular site. It is therefore crucial that research and monitoring is built into prescription burning programs in each area (Christensen and Maisey 1987; Gill 1989; Whelan and Muston 1991).

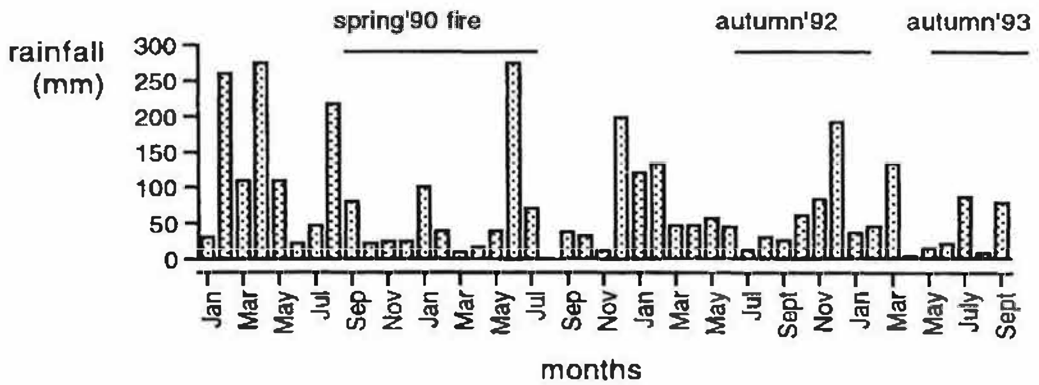

Figure 2. Months to first germination in relation to rainfall. The solid lines above the rainfall bars represent the time between seed sowing and first germination after the three fires.

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Interdependence of woody plants, higher fungi and small marsupials in the context of fire

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ABSTRACT

As a 'fire weed', the gastrolobiums and other mycorrhizal peas form dense thickets after intense fires: these serve as essential shelters and nesting sites for the woylie, *Bettongia penicillata*. This marsupial is the major consumer of the underground sporocarps of at least 18 species of higher fungi in eucalypt forest in Western Australia. When fresh faecal pellets from captured animals were applied to seedlings of *Gastrolobium bilobum* (Fabaceae) and *Eucalyptus calophylla* (Myrtaceae) in autoclaved soil they formed far more ectomycorrhizal rootlets than the controls in non-autoclaved soil. Application of fresh spores of two hypogeous species to the seedlings produced neither mycorrhizas nor growth responses. The most likely explanation is that digestion by the marsupial facilitates germination of the spores. More recent work on the *Gastrolobium* has shown the pellets are likely to increase the numbers and frequency of mycorrhizal types from one or two to five in soils varying in fire history. Restoration of these fungi after fire appears vital for re-establishment of species such as the *Gastrolobium* as it is killed by fire. Not only does consumption of sporocarps escalate after fire but the woylie may travel up to 3 km overnight and moves from one burnt patch to another. This gives it the capacity to be an effective dispersal and restoration agent, but the extent to which the soil is sterilized by fire remains uncertain.

INTRODUCTION

Fires have direct and indirect effects on the ecosystem. My aim here is to highlight one of the intriguing indirect effects, that of the relationships between major woody plants, higher fungi and certain moderately small marsupials. While the full details are still under study or remain unexplored, it is already clear that this triangular trophic relationship is a prime example of the 'balance of nature'. The starting point for any species conservation or habitat management program is to understand the repercussions of any change in status of one component on other components of the ecosystem. I focus on the southern dry sclerophyll forest in Western Australia as the system I know best, referring to work in eastern Australia where this helps amplify points. I examine the interacting components in turn with fire as the starting point for restoration of the ecosystem. I concentrate on fungi which form sheathing (ecto) mycorrhizas with the rootlets of many plants in Australian forests (Fig. 1, Warcup 1980a).

RESPONSES TO FIRE

Effect of Fire on Mycorrhizal Fungi

The effect of fire on mycorrhizal fungi depends on where the mycelium and mycorrhizas are located in relation to the intensity of the fire. Mycorrhizal fungi live saprophytically in litter or on rootlets in decaying litter or the organically rich surface soil (Malajczuk and Hingston 1981; Reddell and Malajczuk 1984). Intense fires will burn off the litter and heat the soil to a depth of 100 mm or more. Malajczuk and Hingston (1981) showed the density of mycorrhizal rootlets was lower after an intense fire than the controls in *Eucalyptus marginata* (jarrah) forest. A decrease in rootlet numbers could have contributed to these results and no counts were made before the fire. Subsequent work (Reddell and Malajczuk 1984) showed that prescribed burning reduced by 90 per cent the numbers of white and brown ectomycorrhizas in *E. marginata* but not the deeper located black mycorrhizas. Warcup (1981) noted that a steam treatment of 55°C for 5 minutes was sufficient to reduce fungal numbers markedly. While all *E. regnans* seedlings grown in soils heated at 55-82°C for 5 to 30 minutes were mycorrhizal, several ectomycorrhizal types formed by basidiomycetes present in the controls were absent.

In contrast to the above, there are many instances of the stimulation of sporulation by fire. This is most obvious for the epigeous (toadstool-type) fungi (Christensen 1980), many of which are

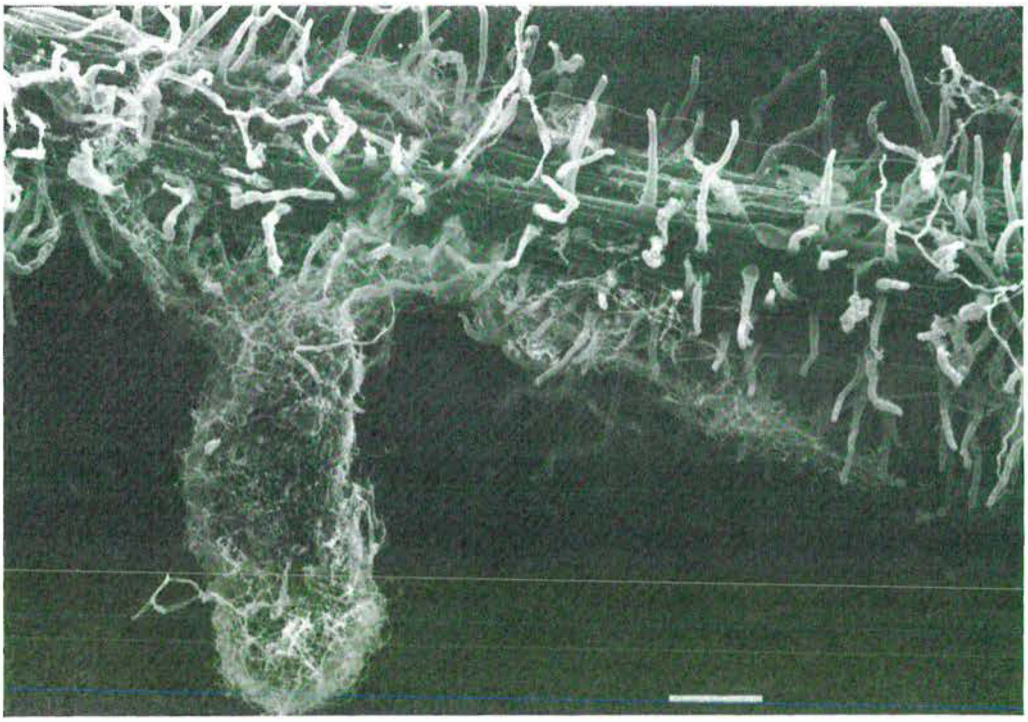

Figure 1. Scanning electron micrograph of mycorrhiza rootlet (simple, tan type) on Gastrolobium calycinum *from Dryandra State Forest. Scale = 0.1 mm.*

ectomycorrhizal, in the autumn after fire. The hypogeous (underground puffball-type) sporocarps are much more difficult to observe (Fig. 2). After a wildfire, most hypogeous discomycetes only sporulated in the first year, nine of which formed ectomycorrhizas with *Eucalyptus obliqua* or *Melaleuca uncinata* (Warcup 1990). In lighter burnt areas, hyphae grew up from the non-heated part of the surface soil and produced abundant hypogeous basidiocarps in the second winter-spring.

Johnson (in Taylor 1992a) claims that sporulation may be much quicker than this - at least in Tasmania: sporocarps were more common a week after a burn than in the controls, rising to six times greater eventually. This appears to be a specific enhancement of the hypogeous genus *Mesophellia* (Taylor 1991), a major ectomycorrhizal fungus in Australia because of its association with eucalypts (Ashton 1976; Malajczuk *et al.* 1987; Dell *et al.* 1990). Johnson's research was in response to suggestions that increase in sporulation was more apparent than real, as it is easier to locate *Mesophellia* sporocarps after fire and they are more likely to survive than other groups (Christensen 1980; Warcup 1990; Claridge 1992), but it has yet to be fully reported. The formation of discomycete mycorrhizas on several eucalypts was stimulated by steam heating soil at 55-82°C for up to 30 minutes (Warcup 1981) - presumably germination of the spores was stimulated by the heat.

Effect of Fire on Mycophagous Marsupials

The woylie *(Bettongia penicillata)* is the main consumer of fungi (mycophagy) in Western Australia (Christensen 1980). The short-nosed bandicoot *(Isoodon obesulus)* and the southern bush rat *(Rattus fuscipes)* occasionally dig up and consume sporocarps. Within six days of a prescribed burn, there were more excavations per unit area in burnt patches than in the controls (Table 1). Most (73 per cent) of the holes in burnt areas were surrounded by *Mesophellia* remnants, while there was no evidence of fungi for 97 per cent of holes in the unburnt areas: either a different (softer) type of sporocarp which is more edible predominates in the unburnt sites (Claridge *et al.* 1993) or many diggings yielded no sporocarps. The latter is less likely as the animals rely on the odour of sporocarps to locate them. Over a two-year period since fire, there were 10 ± 9 (sd) woylie holes per 100 m^2 in the burnt patches and 3 ± 2 in the unburnt (Lamont *et al.* 1985). A similar pattern has been noted for *Bettongia gaimardi* in Tasmania (Taylor 1991) and the potoroos (*Potorous longipes* and *P. tridactylus*) in south-eastern Australia (Guiler 1971; Bennett and Henry in Claridge 1992).

Effect of Fire on Woody Plants

Some plant species survive fire (resprouters) while others are usually killed (non-sprouters). Most trees in dry sclerophyll forest recover but their non-lignotuberous seedlings are killed (Abbott and Loneragan 1984). More important is the diverse array of sclerophyllous shrubs in the understorey. Although many of these are killed, germination of their seeds is stimulated by fire heat - either directly through

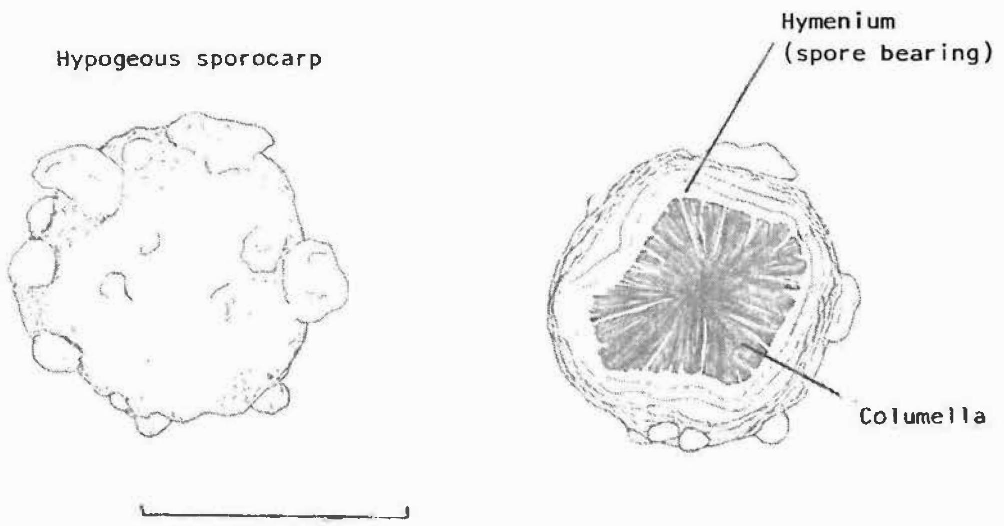

Figure 2. One of the two most abundant types of underground (hypogeous) sporocarp in gully vegetation at Perup forest. Note the lateritic pebbles, charcoal and sand embedded in the outer peridium on the left, and the cutaway on the right showing the nutritious central columella surrounded by the spore-bearing hymenium. This is a member of the Mesophelliaceae which formed simple, tan-coloured mycorrhizas with potted Gastrolobium bilobum *(N. Malajczuk, pers. comm.). Scale = 20 mm. Drawn by C. Ralph.*

increasing water permeability of their hard soil-stored seeds (Christensen 1980; Warcup 1980b) or indirectly through promoting the release of canopy-stored seeds (Lamont *et al.* 1991). In addition, many of these are ectomycorrhizal (Table 2). Some species are major thicket-formers after fire and contribute greatly to nitrogen fixation (Shea and Kitt 1976) and thus nutrient cycling in the ecosystem (Lamont 1992). There is no doubt that their rapid post-fire growth is largely due to the presence of ectomycorrhizas (Warcup 1980a; Lamont *et al.* 1985).

INTERACTIONS

Relationship between Marsupials and Fungi

Christensen (1980) showed that hypogeous sporocarps are a major part of the diet of the woylie. The nitrogen content of the *Mesophellia* columella (Fig. 2), favoured by mycophagous animals, is comparable to that in green leaves and contains high levels of lipids as well (Kinnear *et al.* 1979). The columella cannot be consumed without ingesting thousands of spores. The number of spore types per monthly batch of woylie faecal pellets was 3-10 (Christensen 1980). Of the 17 spore types identified in faeces, at least 10 belonged to ectomycorrhizal fungal groups (Lamont *et al.* 1985). *Mesophellia* was by far the best represented, especially over summer and after fire, with an average 565 ± 329 (sd) spores per mg pellet. Hypogeous fungi, especially *Mesophellia*, were also the main food of the Tasmanian Bettong, with 8-22 spore types per batch and 49 altogether (Taylor 1992b). Similar sporocarps are important in the diet of the Long-nosed Bandicoot, especially in autumn-winter and after fire (Guiler 1971; Bennett and Baxter 1989). Claridge *et al.* (1992) identified 27 ectomycorrhizal spore types in the scats of this animal.

The idea that these animals act as dispersal agents for these fungi has merit, for how else would spores be dispersed any distance from these underground sporocarps? Mycophagy at least frees the (uneaten) spores from the sporocarps but wind dispersal from the fragments would appear ineffective. The alternative is that most spores are dispersed in the faeces. Although there has been no work on Australian mammals, Cork and Kenagy (1989) showed that hypogeous fungal spores were retained for up to 80 hours, the concentration in faeces peaking at 20 hours, in the squirrel (*Spermophilus saturatus*). Woylies complete a circuit, in and out of burnt areas, up to a distance of 3 km overnight (Christensen 1980). Since sporocarps are highly clumped (Taylor 1992c) and different species (sporocarp types) are associated with different parts of the landscape (Claridge *et al.* 1993), this capacity for extensive transport of spores may be important in ensuring all potential host plants have access to suitable inocula.

The above arguments assume that the spores pass through mammal guts unharmed. Lamont *et al.* (1985) were the first to show this for Australian organisms. Suspensions of macerated faecal pellets collected from caged woylies applied to heat-sterilized Perup soil resulted in the production of five

TABLE 1

Total number of small holes with *hypogeous sporocarp* remains along five transects six days after fire in Perup Forest (from Christensen 1980).

	MESOPHELLIA TYPE	OTHER TYPES	No. SPOROCARPS	TOTAL
Burnt patches	263	19	78	360
Unburnt patches	8	1	285	294

TABLE 2

Selection of sclerophyll shrub species in dry eucalypt forests, most of which are killed by fire, their seed germination is promoted by fire heat and which are also ectomycorrhizal. Collated from Specht *et al.* (1958), Warcup (1980a,b), Ralph (1984), Brundrett and Abbott (1991), D. Bell, J. Warcup and B. Lamont (personal observations).

+ = yes, - = no, ? = uncertain

SPECIES	KILLED BY FIRE	GERMINATION ENHANCED BY FIRE	ECTOMYCORRHIZAL
Acacia myrtifolia	+	+	+
Acacia pycnantha	+	+	+
Bossiaea ornata	-	+	+
Bossiaea prostrata	?	+	+
Chorizema cordatum	+?	+	+
Cryptandra arbutiflora	+?	+?	+
Cryptandra tomentosa	+	+	+
Gastrolobium bilobum	+	+	+
Gastrolobium calycinum	+	+	+
Gastrolobium oxylobiodes	+	+	+?
Gastrolobium spinosum	+	+	+?
Gompholobium marginatum	+	+	+
Gompholobium tomentosum	+	+	+
Gompholobium venustum	+	+	+
Kennedia prostrata	+	+	+
Melaleuca decussata	+?	+	+
Melaleuca unicinata	+?	+	+
Melaleuca viminea	+	+	+?
Mirbelia dilatata	+	+	+
Opercularia varia	+?	+	+
Oxylobium capitatum	-	+	+
Oxylobium lanceolatum	+	+	+
Oxylobium linearifolium	+?	+	+
Pericalymma ellipticum	+	+	+
Platysace heterophylla	-?	+	+
Pultenaea ericifolia	+?	+	+?
Pultenaea scabra	+?	+	+
Pultenaea trinervis	+	+	+
Spyridium cordatum	+	+	+?
Spyridium parvifolium	+?	+	+
Trymalium floribundum	+	+	+
Trymalium ledifolium	+	+	+?

ectomycorrhizal types by six-month-old ***Eucalyptus calophylla*** and five by ***Gastrolobium bilobum***, seven altogether. Plants in heat-sterilized soil receiving sterilized pellets produced no mycorrhizas and grew poorly by comparison (Table 3).

Further, spores removed from the two most common sporocarp types at the site and applied in a similar way yielded no mycorrhizas and even poorer growth. One interpretation is that these fungal species do not form mycorrhizas with these plants. However, one of the inoculated *G. bilobum* plants produced a sporocarp which proved to be the same species as used for inoculation and responsible for one of the mycorrhizal types (simple tan) routinely produced on its root system (Fig. 1). Both plant species had white, pyramidal-type mycorrhizas like those in other eucalypts after inoculation with spores or mycelium of *Mesophellia* spp. (Ashton 1976; Malajczuk *et al.* 1987).

Another possibility is that digestion is required as a pretreatment before germination will occur. This is consistent with the usual failure of ectomycorrhizal fungal spores to germinate in culture or the requirements for special treatments to induce germination (references in Lamont *et al.* 1985). Perhaps reports of success with *Mesophellia* spores actually included active mycelium? The pretreatment hypothesis was supported seven years later by the parallel experiments of Claridge *et al.* (1992). They showed application of spores of the most common species in south-eastern Australian forests, *M. pachythrix*, to two eucalypt species in heat-sterilized soil failed to produce mycorrhizas. They considered that this fungus was responsible for the white, pyramidal-type ectomycorrhizas induced on *E. sieberi* in the presence of scats from the Long-nosed Potoroo, especially as it accounted for 16 per cent of the spores in the scats, as well as non-sterilized soil.

Relationship between Marsupials and Shrubs after Fire

It is possible that the passage of time is sufficient to precondition the spores for germination or that mycelial regrowth after fire is enough for inoculation of new roots of surviving plants or new seedlings. In unpublished work, we examined the effect of scats on mycorrhizal production by *G. bilobum* in Perup soil collected four days after fire (prescribed burn) and from sites burnt 18 months and >10 years earlier. All were fine loams from *G. bilobum* dominated gullies to a depth of 50 mm after brushing away any coarse organic matter. Since woylies began digging within a day of the April 1986 fire, samples were taken away from sites already visited by these animals. By giving the results on a per pot basis, we have an index of how well distributed was each fungal type in each soil as well as its occurrence at all.

By six months growth, scats had contributed three ectomycorrhizal types and increased the frequency of two others, while a sixth was already present in all pots of the three soils (Table 4). The most notable effects were the addition of two mycorrhizal types to the 4-day burnt soil and two to the 18 month burnt soil. Even in the >10-year burnt soil, inoculation increased the incidence of four ectomycorrhizal types by 16-66 per cent. While the change in mycorrhizal status due to the fresh scats had no effect on plant growth (unpublished), the extra fungal types resulted in a significant increase in the proportion of root tips that were mycorrhizal in the 4-day burnt soil (Table 5). This indicates that scats increase the inoculation capacity of the soil at the level of the individual root system of establishing plants.

In a similar way, Claridge *et al.* (1992) found that scats of the potoroo increased the number of ectomycorrhizal types from five to seven in *E. sieberi*, but the results are less useful as there were no unsterilized soils to which fresh or sterilized scats had been added and the interval since the last fire is not stated. Bougher *et al.* (1990) showed that some isolates and species of ectomycorrhizal fungi are far more effective than others at enhancing growth and P content of *E. diversicolor* at low soil P. By supplementing the range of ectomycorrhizal fungi available at any time the marsupial scats increase the likelihood of the presence of the nutritionally most efficient and the plant can take full advantage of the different metabolic and environmental optima of the various species associated with its root system.

Rapid restoration of the shrub layer is also important for the marsupials. The woylie shelters from predators, nests, feeds, mates and rears its young under the protection of a moderately dense understorey (Christensen 1980; Christensen and Leftwich 1980). It gathers litter to line its nest among branches arising from the ground; and quickly makes other nests nearby after fire. The woylie also collects *G. bilobum* seeds, and occasionally pulls out and consumes young seedlings arising from the caches after fire. While it is one of the most monofluoroacetate-resistant animals known, and *G. bilobum* one of the most toxic (Oliver *et al.* 1977), the poison peas appear to contribute little to the woylie's diet compared with the fungi.

DISCUSSION

There is a remarkable reciprocal relationship between certain small marsupials, shrubs (especially legumes) and ectomycorrhizal fungi (Fig. 3). The partners are different, but the relationships are the same, in eucalypt forests in mainland south-eastern Australia and south-western Tasmania. This partnership is jeopardized by fire; animals scatter or die, plants lose their foliage or die, fungi in the litter or humus are incinerated or heat-killed. However, a series of events is triggered off which soon restores the relationships. The animals return to the burnt sites almost immediately and start searching for the hypogeous sporocarps of ectomycorrhizal fungi, now, if not before, a preferred

TABLE 3

Ectomycorrhizal status and growth of *Eucalyptus calophylla* and *Gastrolobium bilobum* in heat-sterilized soil from their natural habitat (Perup) inoculated with fresh or sterilized woylie scats (faecal pellets) or spores of *Mesophellia trabalis* or *M.* sp. (summarized from Lamont *et al.* 1985).

TREATMENT	*EUCALYPTUS CALOPHYLLA* DRY WEIGHT (mg)	*EUCALYPTUS CALOPHYLLA* MYCORRHIZAL TIPS (%)	*GASTROLOBIUM BILOBUM* DRY WEIGHT (mg)	*GASTROLOBIUM BILOBUM* MYCORRHIZAL TIPS (%)
Fresh scats	1359	56	759	89
Heat-sterilized scats	951	0	81	0
Spores	675	0	50	0

TABLE 4

Percentage of six-month-old plants of *Gastrolobium bilobum* (n = 12-15) bearing each of six ectomycorrhizal types in the presence of fresh or heat sterilized scats of the woylie in three soils of different fire histories. All soils were collected six days after the April 1986 fire (B. Lamont and L. Stewart, unpubl.).

DATE OF LAST FIRE	CONDITION OF APPLIED SCATS	ECTOMYCORRHIZAL TYPE[a] S,W	S,T	S,B	P,T	P,B	C,B
April 1986	fresh	67	100	67	100	0	33
	sterilized	62	100	0	85	0	0
October 1984	fresh	100	100	0	92	58	33
	sterilized	67	100	0	75	0	0
<1976	fresh	83	100	83	92	0	67
	sterilized	67	100	17	67	0	8

[a] S = simple, W = white, T = tan-coloured, B = black, P = pyramidal, C = coralloid

TABLE 5

Growth and ectomycorrhizal status of *Gastrolobium bilobum* grown in April 1986 burn soil. T = results of t - test.

	WOYLIE SCATS FRESH	WOYLIE SCATS STERILIZED	T
Shoot weight (mg)	420±133	402±142	$p > 0.25$
Root tips mycorrhizal (%)	64.9±7.0	57.9±8.3	$p = 0.0001$

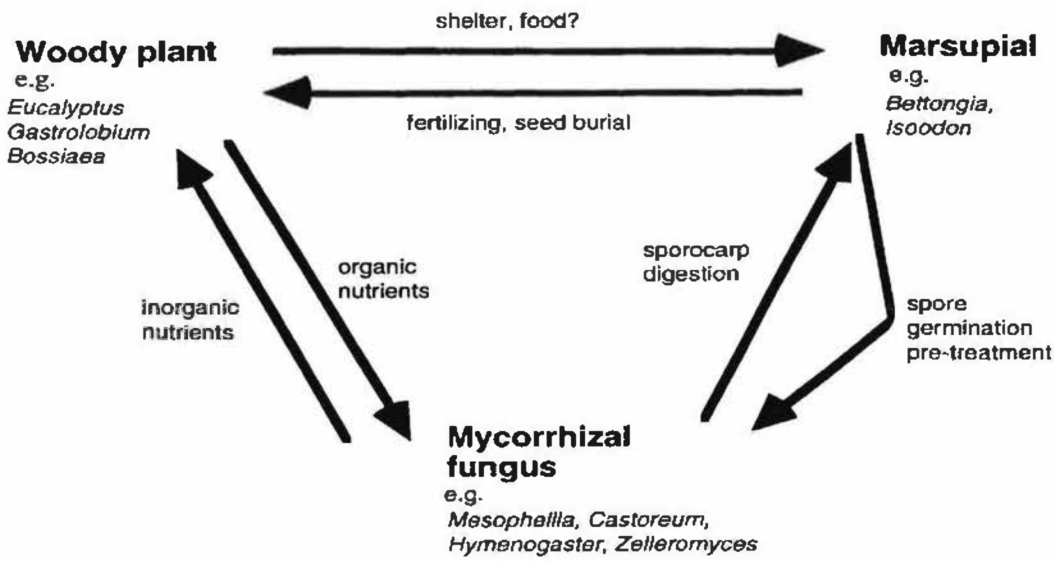

Figure 3. Food web between woody plants (producers), mycorrhizal fungi (heterotrophs), and Bettongia penicillata *and other mycophagous mammals - all relationships are mutualistic and stimulated by fire (modified from Lamont 1994).*

food source. There is some evidence, yet to be fully documented or corroborated, that sporocarp production of the major genus, *Mesophellia*, is stimulated within days of the fire. The marsupials range in and outside the burnt patches redistributing the spores from tight mono-specific clumps to other parts of the landscape.

Unlike parallel partnerships reported elsewhere (e.g. Kotter and Farentinos 1984), there is evidence that spore germination is stimulated by passage through the marsupial's gut for the three species examined so far. Application of faecal pellets increases the number and abundance of ectomycorrhizal fungi in surface soil immediately after fire, but even soils from sites not burnt for more than 10 years may benefit from faecal inocula. Fire predisposes the seeds for mass germination in the first winter after the fire. Early ectomycorrhizal formation ensures rapid growth and guarantees establishment of a moderately dense understorey in a few years. In the prolonged absence of fire (>25 years, Christensen 1980), the partnership is again in crisis, as non-sprouting shrubs die, the vegetation becomes very open, and presumably sporocarp production declines.

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Burning grevilleas, ants, rats and wallabies

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ABSTRACT

Many *Grevillea* spp. in eastern Australia are killed by fire and rely on germination from a soil seedbank to re-establish populations after fire. The magnitude and build-up of this soil seedbank will strongly influence post-fire recovery levels.

Seed predation by native rats and swamp wallabies significantly reduces the size of the soil seedbank in grevilleas. In species where fecundity is low and adults are short-lived, this may severely affect the ability of the species to persist at a site. In such cases, specific management actions are needed to allow fire-free intervals of sufficient length to maximize the magnitude of the soil seedbank. In addition, burns should not be very small in area or seed predation will not be reduced post-fire.

Two ecologically functional seed dispersal syndromes are found in grevilleas in the Sydney region. Species whose seeds lack an elaiosome are not moved by ants. Species that possess an elaiosome are moved by ants. Large ant species that take seeds to nests may be functioning as agents that reduce the impact of seed predators. The importance of such ants depends on a more detailed understanding of the interaction between fire and mammalian seed predator abundance and the fate of seeds moved by ants.

INTRODUCTION

The length of the interval between fires is critical in determining the post-fire abundance of fire-sensitive species. For such species, fires eliminate above-ground plants and the level of recovery depends largely on the magnitude of the soil or canopy seedbank. The length of the fire interval will influence the time seedlings have to mature and replenish the seedbank. The level of post-fire recovery will also be influenced by other factors such as moisture availability, influencing early seedling survival (Bradstock and Bedward 1992) and soil heating controlling post-fire germination in legumes and some other groups (Auld and O'Connell 1991; Auld, Bradstock and Keith unpubl.). However, an understanding of the rate of build-up and magnitude of the seedbank in fire-sensitive species is essential for predicting their response to fires of varying frequency.

Grevillea is a widespread genus in Australia (McGillvray 1993). In the Sydney region of eastern Australia, many *Grevillea* spp. occur in fire-prone communities and are killed by fire, relying on germination from a soil seedbank for post-fire recovery. For such species, fire management should rely on ensuring fire intervals are long enough for seedlings to reach maturity (primary juvenile period, Benson 1985) and a subsequent period for the seedbank in the soil to build-up (Bradstock and Auld 1987). However, no quantitative data on the nature of the soil seedbank in grevilleas exists and it is difficult to accurately predict appropriate fire frequencies for grevilleas. The magnitude and post-fire build-up of a soil seedbank in *Grevillea* spp. will be governed by a range of demographic factors: the primary juvenile period; fecundity; seedling and adult survivorship; predispersal seed predation; seed viability; post-dispersal seed predation; and the dynamics of the soil seedbank itself.

This paper discusses the role that seed dispersal agents and post-dispersal seed predators play in influencing the post-fire build-up and magnitude of soil seedbanks and how this role may be influenced by fire management.

FUNCTIONAL SEED TYPES IN *GREVILLEA*

Two distinct seed types exist in *Grevillea* spp. in the Sydney region. Most species, for example, the widespread *G. buxifolia*, *G. linearifolia* and *G. speciosa*, are myrmecochorous, having seeds with an attached lipid body or elaiosome (Fig. 1). Such seeds attract a range of ant species which commonly move seeds from many plant species in the Sydney region (Rice and Westoby 1981). Ants either remove the elaiosome in pieces *in situ* or, if the ants are large enough, they will try and drag the seeds towards nest entrances and the elaiosome may be removed in the nest (Hughes and Westoby 1992). A few *Grevillea* spp. lack this lipid body (Fig. 2) and seeds are not attractive to ants and have no obvious dispersal mechanisms. These species have toothbrush-type flowers and many are rare or have restricted distributions, e.g. *G. caleyi* and *G. longifolia*.

DISPERSAL AND SEED PREDATION IN *GREVILLEA*

Seeds Lacking an Elaiosome

Seeds lacking an elaiosome are not moved by ants (Auld *et al.* 1993). However, some 82-94 per cent of seeds are consumed on the soil surface by native mammals (bush rats, *Rattus fuscipes* and swamp wallabies, *Wallabia bicolor*, Figs 3a, 4a)(Auld *et al.* 1993). If these mammals are excluded, some seeds are consumed, presumably by insects, although most seeds escape seed predation.

Seeds with an Elaiosome

Two types of seed interaction with ants occur. Small ants such as *Iridomyrmex* and *Pheidole* may move seeds small distances (up to 20 cm), but such ants usually remove pieces of the elaiosome *in situ*. These ants are small compared with the size of grevillea seeds and seeds can only be moved when several ants work together. This is the most common type of ant/seed interaction. Alternatively, large solitary ants such as *Aphaenogaster* and *Rhytidoponera* may be able to drag seeds towards nests (Fig. 3b). Some seeds are discarded *en route* while some presumably are taken into nests, *c.f.* movement of legume seeds by ants (Auld 1986; Hughes and Westoby 1992). The fate of such seeds is unknown, but it is assumed that the elaiosome is removed in the nest and the seed discarded either in the nest or on the surface near the nest (Auld 1986; Hughes and Westoby 1992). At the same time, native rats and wallabies also eat seeds on the soil surface (Figs 3b, 4b) and a small amount of seeds are lost to insect predators (Auld *et al.* 1993).

The proportion of seeds moved away from mammalian seed predators by large ant species is difficult to estimate since the fates of seeds in nests of these ants is unknown. Seeds may be ejected from nests once the elaiosome is removed (Hughes and Westoby 1992) and they then would be subjected to mammal seed predation. Alternatively, seeds may be discarded in chambers in the soil (Auld 1986). If it is assumed that all seeds encountered by large ants escape mammalian seed predation then, up to 20 per cent of seeds may avoid mammal predation in this manner (Fig. 3b). In reality, seeds moved by large ants may be dropped prior to reaching nests, discarded on the soil surface, buried too deeply for subsequent emergence or buried in 'unsafe sites'. A study of the fates of seeds moved by ants is needed to quantify the proportion of seeds that may escape seed predation by being moved by ants.

MODELS OF THE REPLENISHMENT OF SOIL SEEDBANKS AFTER FIRE IN *GREVILLEA*

The impact of ants, rats and wallabies on the soil seedbank can be modelled using demographic data for grevilleas (Auld *et al.* 1993; Auld unpubl.). These models incorporate data on plant survival, fecundity, pre- and post-dispersal seed predation, seed viability and seed longevity in the soil. Three alternative scenarios are presented, with all models representing the number of seeds accumulating in the soil seedbank for each seedling that emerges after a fire at time zero. The residual seedbank that does not germinate after any one fire is not included.

(1) seed predation by mammals is constant (82-94 per cent) throughout all post-fire fruiting years. This varies slightly between sites and is based on data from populations that were unburnt for 12-18 years (Auld *et al.* 1993).

(2) no seed predation by mammals. Although escape from mammalian seed predation has not been observed in the field (Auld *et al.* 1993), this scenario allows a consideration of the impact that seed predators (scenario 1) have on the build-up of a soil seedbank after fire.

(3) seed predation by mammals varies as the density of mammals varies post-fire. Data from Fox and McKay (1981) on changes in post-fire numbers of *Rattus fuscipes* were used to estimate likely levels of mammalian seed predation post-fire. This scenario assumes that there was no seed predation in the first 5 years after a fire and that predation increased linearly to its maximum level (82-94 per cent) 10 years after fire in response to increasing levels of *Rattus fuscipes*. Although other mammals such as *Mus musculus* and *Pseudomys novaehollandiae* may be common in the first few years post-fire in some habitats (Fox and McKay 1991), there is no evidence that these species eat seeds of *Grevillea* species and no seeds will be produced by *Grevillea* seedlings until they mature some 2-4 years after a fire. The impact of fire on population densities of *Wallabia bicolor* is currently unknown.

Under the scenario whereby mammalian seed predators are present in all post-fire years, these seed predators will have a major impact on the build-up and magnitude of the seedbank in all grevilleas. For two species with no elaiosome (*G. caleyi* and *G. longifolia*) mammals consume most of the annual seed output from plants and only a small proportion of each annual seed-crop is expected to reach the soil seedbank (Fig. 5). Differences between *G. caleyi* and *G. longifolia* reflect differences in fecundity and adult

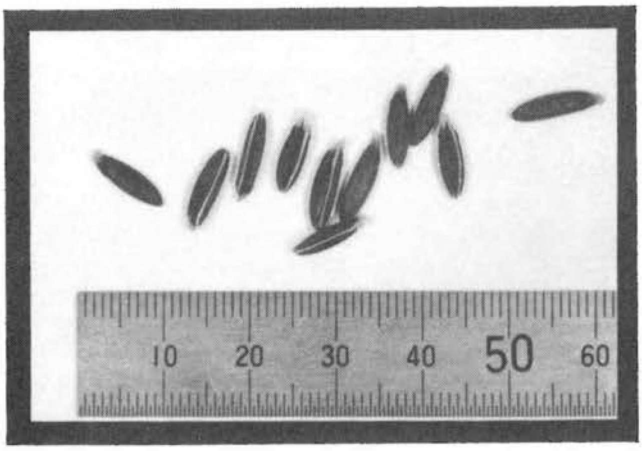

Figure 1. Grevillea sericea *seeds showing the presence of an elaiosome.*

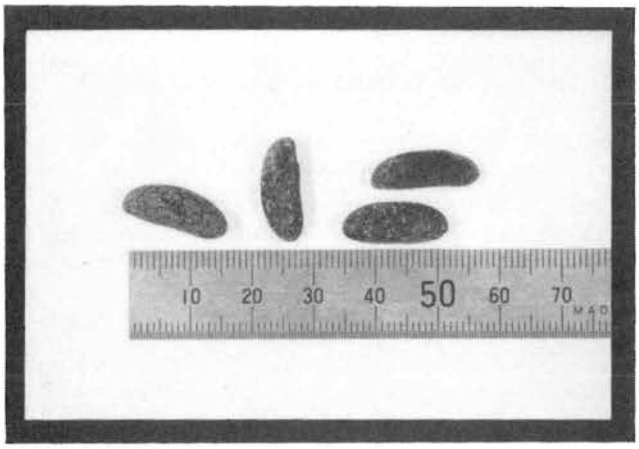

Figure 2. Grevillea caleyi *seeds lacking an elaiosome.*

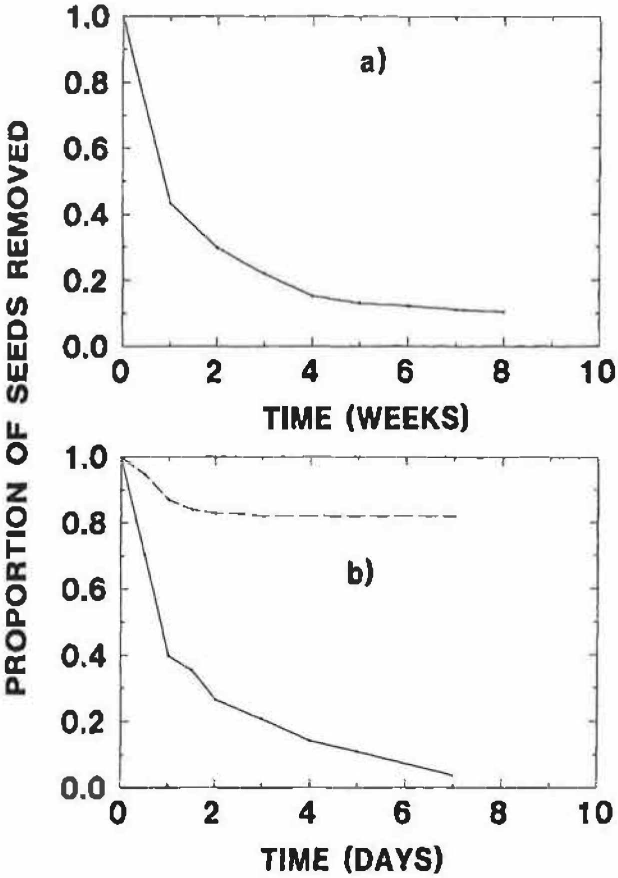

Figure 3. Seed movement and losses during dispersal in Grevillea.

a) grevilleas whose seeds lack an elaiosome: seeds lost to native rats and swamp wallabies.

b) grevilleas whose seeds have an elaiosome.
seeds moved by large ant species. — — — —
seeds moved by all ant species or lost to native rats and swamp wallabies. ————

a) Grevillea caleyi

b) Grevillea buxifolia

Figure 4. Grevillea seeds eaten by bush rats Rattus fuscipes.

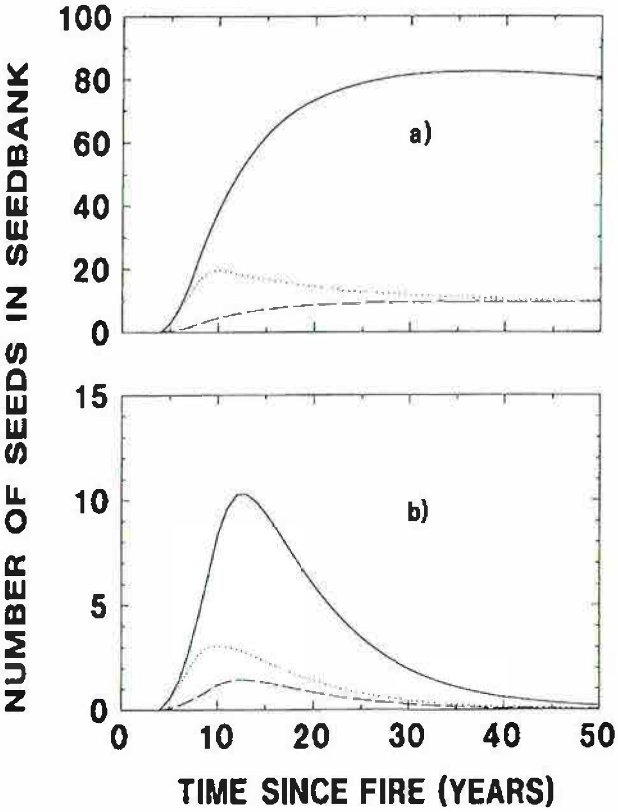

Figure 5. Soil seedbank models for grevilleas whose seeds lack an elaiosome (a) G. longifolia *and* (b) G. caleyi. *Seedbank size represents the number of seeds added to the soil for each emerging seedling after a fire.*

no mammal seed predation. ____________

mammal seed predation reduced in 0-10 years post-fire period in response to reduced mammal abundance (see text) ....

constant mammal seed predation in all post-fire years. — — — — — —

survivorship. Consequently, *G. longifolia* is the more resilient of the two species to mammalian seed predation. The level of seed loss to mammals in the first few fruiting seasons after fire is critical. The seedbank of *G. caleyi* is relatively small and only a small fraction may be long-lived as adult plants senesce in unburnt populations older than 10-15 years. A suitable fire interval for burning populations of this rare plant may have to include both minimum and maximum fire-free intervals. In comparison, the magnitude of the soil seedbank is increased markedly in these grevilleas if mammal seed predation is assumed to be absent in all post-fire years. A similar, but less substantial increase in the seedbank is achieved if mammal seed predation is reduced in the first few post-fire seed-crops (Fig. 5). If mammal seed predators are not reduced in the immediate post-fire environment then the soil seedbank may not be effectively replenished. In the case of *G. caleyi*, the model where mammalian seed predators are always present would predict a limited window of suitable fire-free interval in order to maintain population size and much potential for long-term declines in future population size.

Similar patterns in the three seed predation model scenarios are found for a typical species with an elaiosome, *G. speciosa* (Fig. 6). The overall magnitude of the seedbank is greater in this species than in *G. caleyi* or *G. longifolia* as seed viability is higher, predispersal seed predation is less and there can be up to two seeds per fruit in species whose seeds have elaiosomes. If mammalian seed predation follows the predicted abundance of *Rattus fuscipes* post-fire, i.e., it is low in the first few years post-fire, then the build-up of the soil seedbank is greatly enhanced over the constant mammal predation scenario. This impact is more pronounced in *G. speciosa* than in *G. caleyi* or *G. longifolia* as *G. speciosa* seedlings mature earlier (2 years versus 3-4 years). This escape from seed predation in the first few seed-crops produced after a fire may be one method by which all *Grevillea* spp. re-establish their soil seedbanks post-fire. If mammal seed predators are not reduced in the immediate post-fire environment then the soil seedbank may not be effectively replenished.

If it is assumed that up to 20 per cent of released seeds escape seed predation by mammals through movement by large ant species, the impact of ants on the soil seedbank can be inferred. Where mammal seed predation occurs in all post-fire years (Fig. 7a), the magnitude of the seedbank is markedly increased when large ants are present. Where mammal seed predation is removed in the first few fruit crops and then gradually increases to a constant peak at 10 years post-fire (Fig. 7b), movement of seeds by ants has little effect on the magnitude and build-up of the soil seedbank. Thus, the importance of ants as agents for seed escape from mammalian seed predators will depend on the intensity of mammalian seed predation in the first 10 years post-fire. This intensity will reflect the interaction between fire and the abundance of individuals in the mammal populations themselves.

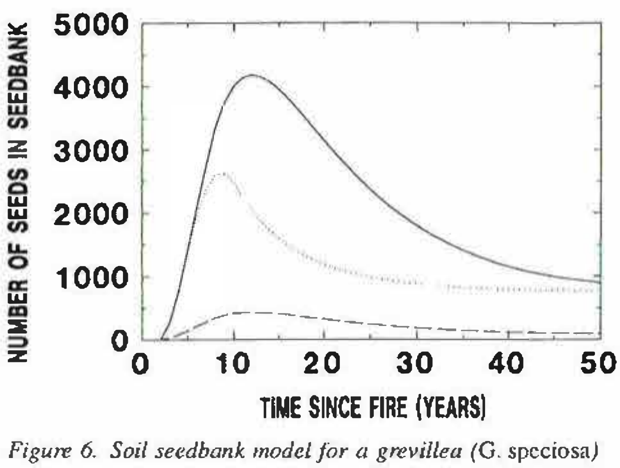

Figure 6. Soil seedbank model for a grevillea (G. speciosa) *with elaiosomes assuming no ant movement of seeds. Seedbank size represents the number of seeds added to the soil for each emerging seedling after a fire. Lines follow Figure 5.*

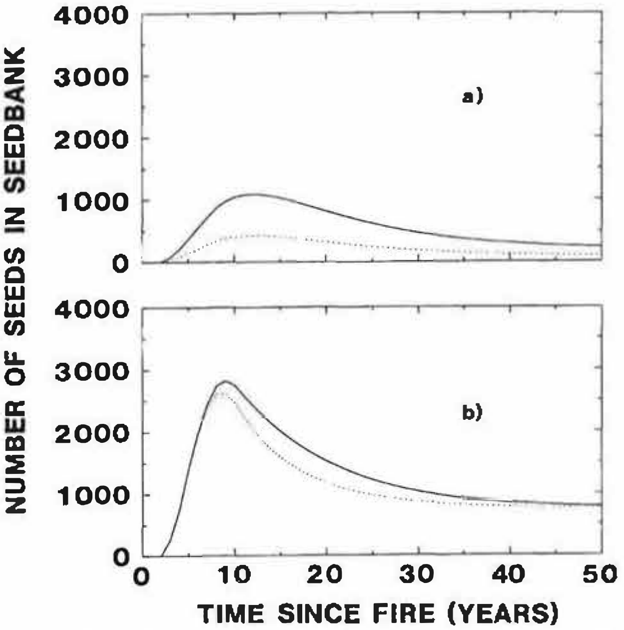

Figure 7. Soil seedbank model for a grevillea (G. speciosa) *with elaiosomes allowing for some seed escape via movement by large ant species.*

(a) constant mammal seed predation in all post-fire years.
large ants move some seed to safe sites; ___________
no seed movement by ants. ....

(b) mammal seed predation reduced in 0-10 years post-fire period in response to reduced mammal abundance (see text).
large ants move some seed to safe sites; ___________
no seed movement by ants. ....

CONCLUSIONS

Native mammals can have a major impact on the post-fire development of soil seedbanks in grevilleas. The interaction between fire and populations of the mammals, in terms of post-fire mammal abundance, has important consequences for grevilleas. This is especially true for rare, relatively short-lived species, with low fecundity. One such species is *G. caleyi* and management of this species needs to address both minimum and maximum fire-free intervals. This grevillea is restricted to several small populations in the Terry Hills areas in the northern suburbs of Sydney. Populations of this species have previously been burnt by small scale burns, 20 x 20 m and 50 x 50 m. Such burning is inappropriate for this grevillea as populations of mammalian seed predators are not reduced in the post-fire environment as they survive in the large surrounding unburnt areas. Consequently, the relationship between the size of an area burnt and post-fire mammal abundance needs further investigation.

For those grevilleas whose seeds possess elaiosomes, ants may function to reduce the impact of mammalian seed predators and hence, to increase the magnitude of soil seedbanks. However, this remains to be clearly demonstrated. If all large ant species simply eject seeds from nests then seeds can still be consumed by mammals. Models of the soil seedbank of grevilleas with elaiosomes suggest that the movement of seeds by ants will be most important when seed predation by mammals is high in the first few seed-crops produced after a fire. If seed predation by mammals is low in this period, seed movement by ants is relatively unimportant to the build-up of a soil seedbank.

For all grevilleas, the level of recruitment after a fire will be largely influenced by the interaction between fire and mammalian seed predators, while ants may also play a role where seeds possess an elaiosome.

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Fire and invertebrates - a review of research methodology and the predictability of post-fire response patterns

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ABSTRACT

Invertebrates are now recognized as critical elements in the maintenance of ecosystems, and many are seen to have potential as bio-indicators of environmental conditions. Information on their responses to environmental disturbance is therefore critical. With regard to the impact of fire on invertebrates, however, many conflicting results have arisen. It was therefore considered timely to carry out a detailed review of the sampling methods employed in most of the studies to date, and of the post-fire invertebrate response patterns recorded.

Considerable variability was found in the sampling methods used, and there were many shortcomings in experimental design and length of study, with few studies adopting an experimental approach incorporating adequate pre- and post-fire sampling. Such deficiencies make it difficult to determine whether the outcomes observed are a true feature of invertebrate responses to fire, or are largely artefacts of the sampling procedure.

Available data indicate that Araneae and also probably Lepidoptera, Isopoda, Blattodea and Thysanura are sensitive to fire and exhibit consistent response patterns across a variety of habitat types, thus qualifying them as potential indicator groups. Trends in invertebrate resilience to fire across broad habitat and climatic gradients suggest there is a need for conservatism in the application of high frequency and large scale fire regimes, particularly in the more mesic forested areas of Australia.

INTRODUCTION

The role of prescribed burning in the management of vegetation throughout temperate Australia is a complex and sometimes contentious issue. Currently, most fire management decisions are based on a reasonably detailed knowledge of weather, fuel and other site parameters and their influence on fire behaviour. This knowledge forms the main rationale for the prescribed burning of forested areas of south-western and south-eastern Australia to reduce fuel loads and wildfire hazard (Shea *et al.* 1981; Cheney 1985; McCaw and Burrows 1989).

In the context of ecosystem management, however, most decisions must be made against a background of little research data on the effects of fire (or a fire regime) on the biota. There is thus a serious dichotomy in our levels of understanding of the principles of fire physics and prescibed burning technology on the one hand, and the impacts of fire and its role in ecosystem management on the other. This points to an urgent need to obtain reliable data on the effects of fires of varying intensities and season of burn on the biota, in conjunction with studies on the behaviour of these fires and the management planning processes involved.

Most studies of fire effects have concentrated on vegetational aspects, perhaps because plants are the organisms perceived as being most directly affected by fire. Until the last decade, relatively little work had been carried out relating to the effects of fire on fauna, and there is still a dearth of knowledge with respect to herpetofauna and invertebrates (see reviews by Suckling and Macfarlane 1984; Christensen and Abbott 1989; Friend 1993).

Invertebrates as Bio-indicators

There is a considerable and growing body of evidence that invertebrates are more important in the maintenance of ecosystems than are vertebrates, yet there remains a paucity of information on them (Key 1978; Majer 1987; Hill and Michaelis 1988). Most of the biological diversity we are dealing with in nature conservation is contributed by invertebrates (Greenslade and Greenslade 1984; New 1984, 1987; CONCOM 1989; Kim 1993), but they are usually ignored in ecological research (Majer 1987). Furthermore, certain guilds of invertebrates are proving excellent bio-indicators of environmental conditions, including pyric status. Groups which have received some attention to date include ants (Majer 1983; Andersen 1987), spiders (Clausen 1986; Main 1987) and beetles (Friend and Williams 1993). Through analysis of insects comprising small vertebrate diets (e.g. of dasyurids, lizards, frogs) and insect/plant interactions (especially herbivory rates), invertebrate studies have the potential to contribute to an understanding of the processes involved in pyric disturbance ecology.

Invertebrates and Fire

As part of a research program examining the impact of experimental fires on invertebrate communities within remnant shrublands in the Stirling Range National Park (Friend and Williams 1993), a significant portion of the literature on fire and invertebrates was reviewed. From this general review it became apparent that most impacts are relatively short-term (e.g. less than 2-3 years), that high intensity wildfires have much greater impacts than lower-intensity prescribed burns, and that spring prescribed burns may have a greater impact than those carried out in autumn. The review, however, also highlighted that a wide variety of post-fire response patterns may occur, and that these are often not consistent within taxonomic group or habitat type between different studies. In many instances invertebrate groups show marked locality, season and year-to-year effects which outweigh any changes attributable to fire.

Furthermore, many inconsistencies seem to have arisen because of variations or shortcomings in experimental design, taxonomic treatment and length of study. Few studies have any pre-fire data (see review by Majer 1985c for Western Australian studies), or any long-term post-fire data (Abbott 1984; Tap and Whelan 1984; Majer 1980, 1985a, 1985c). In the majority of cases, invertebrates have been identified only to ordinal level, thus potentially masking important changes in species and family composition following fire. In addition, most workers have contemporaneously sampled areas of different fire histories and ascribed faunal differences to the effect of these fires. Given the inherent within-site variability of invertebrate populations (Campbell and Tanton 1981), this assumption of pre-fire homogeneity between control and treatment plots is tenuous. Furthermore, the effects of intensity and season of burning are likely to be profound, but have frequently been ignored.

Given the outcomes which arose from this general overview, it was considered timely to conduct a more detailed examination of invertebrate response patterns following fire in order to: (a) quantitatively assess whether or not various groups (e.g. Orders) have consistent post-fire response patterns; (b) highlight groups or taxa which are sensitive to fire and therefore could serve as indicator species of pyric status; and (c) examine trends in relation to habitat type across a broad climatic gradient. In addition, the opportunity was taken to gather some statistics on experimental design and sampling methodologies employed in studies of invertebrate responses to fire.

STUDIES AND PARAMETERS EXAMINED

Twenty-four studies were reviewed which represented a broad cross-section of research carried out in temperate Australia over the past 40 years. More than half of these studies were conducted during the mid 1980s and a large proportion (62 per cent) were undertaken in Western Australia. However, Victoria, South Australia, New South Wales and the Australian Capital Territory also were represented.

The following parameters were noted for each of the studies reviewed:

- Methods - whether pitfall traps, soil/litter samples combined with heat extraction or hand sort or other method, the preservative used in pits and the number of days the pits were open;
- Duration (months) of pre and post-fire monitoring for experimental studies;
- Duration (months) of post-fire monitoring in studies utilizing a space-for-time approach (Pickett 1989);
- Level of identification - whether order, family or species;
- Habitat type - whether tall open-forest, open-forest, eucalypt woodland, banksia woodland or shrubland;
- Fire type - whether wildfire or prescribed fire;
- Season of fire - whether autumn, spring or summer; and
- Fire intensity - whether high or low;

For each study, the responses of invertebrate groups (mainly orders) were assigned to one of three categories: (a) no marked change in abundance post-fire or highly variable over time (designated zero); (b) a general increase in abundance post-fire (designated plus); and (c) a general decrease in abundance post-fire (designated minus). In this manner response data were accumulated for a total of 20 invertebrate groups (mainly orders) *viz.* Oligochaeta (earthworms), Araneae (spiders), Pseudoscorpionida (pseudoscorpions), Acarina (mites), Isopoda (slaters), Chilopoda (centipedes), Diplopoda (millipedes), Collembola (springtails), Thysanura (bristletails), Blattodea (cockroaches), Isoptera (termites), Dermaptera (earwigs), Orthoptera (grasshoppers), Hemiptera (bugs), Thysanoptera (thrips), Coleoptera (beetles), Diptera (flies), Lepidoptera (moths and butterflies), Hymenoptera (bees and wasps but excluding ants), and Hymenoptera (ants).

RESULTS AND DISCUSSION

Sampling Methods

Seventeen of the 24 studies reviewed (71 per cent) used pitfall traps, but the dimensions varied greatly and were often tailored for specific purposes. For example, small diameter pitfall traps (e.g. 18 mm test tubes) have frequently been used by researchers with a primary interest in ants (e.g. Majer 1980), while more generalized studies have utilized larger traps (e.g. plastic vials or cups up to 90 mm diameter; Abbott 1984; Friend and Williams 1993; Strehlow 1993). One study (Campbell and Tanton 1981) used pitfall traps but provided no details of their dimensions or preservative used, while Bornemissza (1969; abstract only) did not describe any of the methods used in his study.

A mixture of ethanol and glycerol was the most common preservative used in pitfall traps (10 studies), followed by Galt's solution (Friend and Williams 1993; Main and Gaull 1993; Strehlow 1993) and methanol (Neumann and Tolhurst 1991; Neumann 1991). Pitfall traps were left open for between two to 14 days, with the majority (9 of the 13 which provided such detail) being either seven or ten days.

Heat extraction of soil cores and/or leaf litter samples using Tullgren or Berlese funnels was also a common method used (11 studies) while two studies (Springett 1976, 1979) employed hand sorting. Clearly, as pointed out by Campbell and Tanton (1981), this latter method would be biased against small, cryptic animals and could not be recommended for general studies of invertebrate communities.

Understandably, most researchers use their own method consistently, and while this makes studies by the one person comparable, the large differences in trap sizes, layout, days of sampling and preservative used render quantitative comparisons between studies invalid. Indeed, in examining the many variables affecting sampling with pitfall traps, Adis (1979) called for the quantitative testing of many existing designs and the development of a standard pitfall trap for future universal use. However, trends in the abundance of invertebrate groups following disturbance can be compared for different studies, and this was the approach adopted here.

Duration of Monitoring

Eighteen of the 24 studies examined were based on sampling before and after fire, although two studies (Bornemissza 1969; Hutson and Kirkby 1985; both in summary form) did not specify the length of post-fire sampling. Pre-fire sampling occurred for a mean of 12.2 ± 9.3 months (n = 18; range 1-36; CV = 76 per cent), while post-fire monitoring proceeded for a mean of 18.5 ± 11.2 months (n = 16; range 1-39; CV = 60 per cent). Nine studies (50 per cent) had less than 12 months pre-fire data, but 13 (72 per cent) had more than 12 months post-fire data. Thus many studies had acquired only a minimal amount of pre-fire data, but had adequate post-fire data. Indeed, five studies (Leonard 1972; Abbott 1984; Majer 1984; Andersen 1988; Neumann and Tolhurst 1991) showed great discrepancies in the duration of pre- and post-fire monitoring, with all but Andersen (1988) having less than seven months pre-fire data but one to three or more years of post-fire information. In essence, those studies with the most post-fire data were also among those with the least pre-fire data. Such an unbalanced sampling schedule is of questionable benefit.

Nine studies adopted a space-for-time approach either wholly (i.e. with no pre-fire information; McNamara 1955; Springett 1976, 1979; Whelan *et al.* 1980; Abbott, van Heurck and Wong 1984; Majer 1985b) or as an adjunct to before/after experimental work (e.g. Bamford 1986; Strehlow 1993; Friend and Williams 1993). The latter approach has the advantage in providing both short-term and long-term data on the impact of several fires within a relatively short research time-frame. Space-for-time monitoring proceeded for a mean of 8.4 ± 12.2 months (n = 8, [McNamara (1955) gave no details] range 0.3 - 36 months), but duration was highly variable (CV = 145 per cent).

Level of Identification

The majority of studies (about 75 per cent) examined one or two groups in detail to the family or species level and identified the remainder to order level. The family or species level identifications generally concerned ants and to a lesser extent spiders. Five studies (Curry *et al.* 1985; Hutson and Kirkby 1985; Andersen 1988; Main and Gaull 1993; Strehlow 1993) concentrated only on specific groups (ants, spiders, mites or springtails) and did not examine other order level data. Thus studies have been either very general or highly specific, and to date no fire ecology study has identified a broad range of invertebrates to family or species level. This situation is understandable given the huge diversity of invertebrates and the current (and specialized) taxonomic knowledge of this fauna (the 'taxonomic impediment', New 1984), and makes the search for species or groups which can be used to indicate certain environmental regimes all the more pressing (New 1984). The newly developing field of Rapid Biodiversity Assessment (RBA; Beattie *et al.* 1993) also shows much promise in helping to resolve the taxonomic problems associated with the use of invertebrates in ecological studies.

Research Emphasis to Date

In order to gain some insight into where the emphasis has to date been directed in invertebrate fire ecology research the 24 studies were each categorized according to habitat type, fire type, season and intensity. Classifying studies on this basis immediately indicates where the major research emphasis has been to date, and where more effort is needed.

Only four of the 24 studies reviewed addressed tall open-forest, and only Neumann (1991) examined wildfire impacts, this being for a high intensity summer burn. The remaining three studies addressed prescribed burns, two being high intensity summer/autumn burns (O'Dowd and Gill 1985; Curry *et al.* 1985) in *Eucalyptus delegatensis* and *Eucalyptus diversicolor*, and the other a low intensity spring burn in *E. diversicolor* (Springett 1976).

Eleven of the reviewed studies addressed open-forest, but only that by Hutson and Kirkby (1985) focussed on wildfire, this being a high intensity autumn burn. Of the remaining 10 studies in this habitat type which addressed prescribed fires, four examined autumn burns (Springett 1979; Abbott 1984; Majer 1984; O'Dowd 1985; Neumann and Tolhurst 1991), five examined spring burns (McNamara 1955; Leonard 1972; Springett 1976; Campbell and Tanton 1981; Abbott, van Heurck and Wong 1984), while the comprehensive studies of Neumann and Tolhurst (1991) examined both spring and autumn burns. All except the case studied by Springett (1976) were low intensity prescribed burns.

Three of the reviewed studies encompassed eucalypt woodland, all focussing on low intensity spring burns (Majer 1980, 1985b; Andersen 1988), but with the space-for-time study of Majer (1985b) also examining impacts of a low intensity autumn fire in wandoo woodland. Banksia woodland was also examined by three studies, one investigating a high intensity autumn wildfire (Whelan *et al.* 1980), another comparing moderate to high intensity spring and autumn prescribed burns (Bamford 1986), and the third not specifying fire season or intensity (Bornemissza 1969).

Shrublands were investigated in four of the reviewed studies, three of these having recently been carried out in the wheatbelt and south coast areas of Western Australia. Two of the studies concerned high intensity summer/autumn wildfires (Tap and Whelan 1984; Main and Gaull 1993), one focussed on a high intensity autumn burn on an isolated wheatbelt nature reserve (Strehlow 1993), while the studies of Friend and Williams (1993) examined the relative impacts of low-moderate intensity spring and autumn burns and a high intensity autumn wildfire.

From the above it is clear that most of the emphasis in invertebrate fire ecology has been on prescribed burning in open forest, an area of considerable controversy Australia-wide. Although such effort is well placed, there has been little concensus reached from the results to date (e.g. Campbell and Tanton 1981; Abbott 1984; Majer 1984; *cf* McNamara 1955; Springett 1976, 1979) and other areas, particularly the drier woodlands and shrublands, have received comparatively little attention. There is a need for comprehensive long-term research to clarify the impacts of fire in forest environments, but there is also clearly a need to gather more data from the drier ecosystems and to commence work in the mulga woodlands and hummock grasslands which have received no attention to date. Such studies are now underway in Western Australia (A. Start, S. van Leeuwen, D. Pearson *personal communications*).

Invertebrate Response Patterns and Indicator Groups

Response patterns of the major invertebrate groups were examined across all 24 studies, and with the studies separated according to major habitat type. For the latter analysis there were insufficient data to examine tall open forest (only four studies) and the data for the drier woodland and shrubland habitats were combined. For each invertebrate group for which post-fire response information was available, the proportion of cases showing no post-fire change in abundance (designated zero), a post-fire increase (plus) or a post-fire decrease (minus) was calculated and tabulated. This provided information on both the strength and the consistency of the post-fire responses which could be compared between different invertebrate groups or between the same groups in different habitat types. Thus, in highlighting potential indicator groups, those which are common across a broad range of habitats, and score only in the minus or the zero and minus categories (and not those which score in both the plus and minus categories) should be considered. Groups which satisfy these criteria are common, sensitive to fire and show a consistent response to it, thus conforming to the desirable criteria for selection of indicator taxa (New 1984). A further restriction is that there must be an adequate number of cases (studies) which have provided the data for the particular group (e.g. >5 cases).

Indicator Groups

Across all studies (Table 1) six invertebrate groups satisfy these criteria: Araneae, Isopoda, Thysanura, Blattodea, Isoptera and Lepidoptera. A seventh group, the Diptera, may also qualify because the two plus response cases both related to high intensity wildfires (Neumann 1991; Friend and Williams 1993), indicating some differential but consistent responses associated with type of fire. All of the remaining (13) groups were either insufficiently studied or were inconsistent in their responses.

In open-forest (Table 2) Araneae, Isopoda, Blattodea, Diptera and Lepidoptera were again quite consistent, and, in addition, Acarina, Collembola and Coleoptera qualified. Although few studies included Thysanura (4), all responses were minus suggesting that this group also may be worthy of consideration in this habitat type. Interestingly, ants, which have been frequently used as indicator species registered inconsistent responses in this habitat type. In the drier woodland and shrubland habitats, however, (Table 3) ants showed very consistent post-fire responses (increases), suggesting that their value as indicators of disturbance or pyric status may vary according to climate and habitat.

In the woodland/shrubland habitats (Table 3) there are a paucity of data, but one group stands out as a potential indicator: Araneae. This is not surprising considering that spiders are at the apex of the invertebrate food pyramid, and some representatives (especially the mygalomorphs or trapdoor spiders) are relictual in their distribution, are long-lived and relatively sedentary with poor dispersal powers, and

TABLE 1

Overall post-fire response patterns for major invertebrate groups

GROUP	TOTAL # STUDIES	PROPORTION IN CATEGORIES		
		0	+	−
Acarina	14	0.50	0.10	0.40
Araneae	21	0.60		0.40
Blattodea	11	0.40		0.60
Chilopoda	8	0.50	0.25	0.25
Coleoptera	18	0.50	0.10	0.40
Collembola	13	0.50	0.10	0.40
Dermaptera	4	0.75		0.25
Diplopoda	10	0.30	0.30	0.40
Diptera	12	0.40	0.20	0.40
Hemiptera	13	0.20	0.50	0.30
Hymenoptera (ants)	15	0.20	0.60	0.20
Hymenoptera (excl. ants)	8	0.25	0.25	0.50
Isopoda	8	0.40		0.60
Isoptera	7	0.40		0.60
Lepidoptera	7	0.30		0.70
Oligochaeta	2	0.50		0.50
Orthoptera	10	0.60	0.10	0.30
Pseudoscorpionida	3	0.30		0.70
Thysanoptera	3			1.00
Thysanura	6	0.30		0.70
Total Number of Groups in Categories		19	10	20

0 = no change; + = increase; − = decrease

TABLE 2

Post-fire response patterns for major invertebrate groups
in open forest.

GROUP	TOTAL # STUDIES	PROPORTION IN CATEGORIES		
		0	+	−
Acarina	6	0.50		0.50
Araneae	6	0.70		0.30
Blattodea	5	0.20		0.80
Chilopoda	3	0.70		0.30
Coleoptera	8	0.50		0.50
Collembola	5	0.40		0.60
Dermaptera	2	1.00		
Diplopoda	5	0.60	0.20	0.20
Diptera	6	0.70		0.30
Hemiptera	4	0.50		0.50
Hymenoptera (ants)	6	0.50	0.20	0.30
Hymenoptera (excl. ants)	3	0.30		0.70
Isopoda	5	0.40		0.60
Isoptera	4	0.75		0.25
Lepidoptera	5	0.40		0.60
Oligochaeta	2	0.50		0.50
Orthoptera	4	0.25	0.25	0.50
Pseudoscorpionida	2	0.50		0.50
Thysanoptera	2			1.00
Thysanura	4			1.00
Total Number of Groups in Categories		18	3	19

0 = no change; + = increase; − = decrease

TABLE 3

Post-fire response patterns for major invertebrate groups in woodland/shrubland.

GROUP	TOTAL # STUDIES	PROPORTION IN CATEGORIES		
		0	+	–
Acarina	5	0.60	0.20	0.20
Araneae	11	0.60		0.40
Blattodea	4	0.75		0.25
Chilopoda	3	0.30	0.70	
Coleoptera	7	0.70	0.15	0.15
Collembola	6	0.30	0.30	0.30
Dermaptera	1	1.00		
Diplopoda	2		1.00	
Diptera	4		0.50	0.50
Hemiptera	7	0.15	0.70	0.15
Hymenoptera (ants)	6		1.00	
Hymenoptera (excl. ants)	4	0.25	0.25	0.50
Isopoda	0			
Isoptera	2			1.00
Lepidoptera	1			1.00
Oligochaeta	0			
Orthoptera	5	0.80		0.20
Pseudoscorpionida	0			
Thysanoptera	0			
Thysanura	2			1.00
Total Number of Groups in Categories		10	9	12

0 = no change; + = increase; – = decrease

have very specific microhabitat preferences (Main 1987). Araneae would thus appear the most promising invertebrate indicator group to use in fire ecology studies across a broad range of habitats. Isopoda, Blattodea, Lepidoptera and perhaps Thysanura also appear to have potential as indicator groups, but more data are needed to confirm this. Many of the larger groups for which data are adequate (e.g. Acarina, Collembola, Isoptera, Coleoptera, Diptera and Hymenoptera (ants)) show consistent patterns only in certain habitat types and cannot therefore be considered good indicator groups in fire ecology studies. This is not to say, however, that certain *species* or *taxa* within these groups may not be good indicators, satisfying the criteria defined by New (1984). At this stage, however, there are insufficient fire ecology data available at the species level (perhaps with the exception of ants) to categorize invertebrate species as indicators.

Response Patterns and Habitat Type

Examination of the total numbers of groups in the three response categories for the data in Tables 1-3 indicates there are few cases of post-fire increases in the open-forest. These trends were further investigated by summing the total numbers in the zero (no change), plus (increase) and minus (decrease) categories (irrespective of invertebrate group) for tall open-forest, open-forest and woodland/shrubland. These data revealed very clear and significant differences (X^2 = 32.2 (DF=4); $p < 0.001$) in the response patterns for the three different habitat types (Table 4). In tall open-forest there were more cases than expected in the decrease category, in open forest more cases than expected occurred in the no change and decrease categories, while in woodlands and shrublands the response pattern was similar to that expected by chance.

This outcome, based on the data available to date, strongly suggests that there is a gradient in invertebrate responses to fire related to habitat type, and ultimately, climate. Invertebrates appear to be less resilient (*sensu* Westman 1986) to fire in the more mesic environments (particularly tall open forest) than in the drier woodland and shrubland ecosystems. Such resilience in the drier habitats is probably a reflection of the invertebrate fauna's adaptations to survive seasonal aridity. This does not imply, however, that faunal composition and abundance would not change greatly under a high frequency fire regime in such areas (i.e. exhibit low malleability, *sensu* Westman 1986). Thus in the wetter forested areas, where the invertebrate fauna appears to be even less resilient, significant changes in species

TABLE 4

Total reponses in categories vs habitat type.

	NO CHANGE	INCREASE	DECREASE
Tall Open-forest	7	6	21
	(11.3)	(11.3)	(11.3)
Open-forest	43	3	43
	(29.7)	(29.7)	(29.7)
Woodland/ Shrubland	30	22	18
	(23.3)	(23.3)	(23.3)

Expected values shown in parentheses

abundance and composition may be expected to occur under a high frequency/large scale fire regime.

Clearly, however, we have much to learn before such trends can be verified so that we can confidently devise optimal fire regimes for various habitat types. In particular, we need much more critical and objective data from both the arid areas (e.g. mulga and hummock grasslands) and the forests, and we need to know the relative proportions of fire-sensitive species in the various habitat types. To achieve this, a much more rigorous approach to sampling and experimental design needs to be adopted.

CONCLUSIONS

This detailed review of invertebrate fire ecology studies indicates that a wide variety of sampling methods are employed and that invertebrate response patterns also vary greatly. Given the inherent variability of invertebrate populations it is crucial that studies obtain pre- and post-fire data over several years, and that they are standardized with respect to experimental design and taxonomic treatment. It is only by minimizing/ eliminating such experimental variability that we can determine whether the outcomes observed are a true feature of invertebrate responses to fire, or are largely human-induced. This has important ramifications for the use of invertebrates in studies examining community stability and resilience.

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Responses of reptiles to fire and increasing time after fire in Banksia woodland

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ABSTRACT

In the study of the impact of fire on terrestrial vertebrates in Australian ecosystems, most work has focussed on small mammal species while often richer reptile assemblages have received little attention. Studies on mammals have found that dramatic changes in abundance after fire can be related to changes in vegetation structure; but it is not clear to what extent generalizations based on mammals can be extended to reptiles.

In a three-year study, reptiles were sampled in six areas of *Banksia* woodland unburnt for different lengths of time from 0 to 23 years. The reptile assemblage varied mainly in the first two years after fire, mostly due to differential survival of adult and immature specimens immediately after fire. One species, the small agamid *Tympanocryptis adelaidensis*, was most abundant 3-6 years after fire and this could be related to changes in vegetation structure. Apparent patchiness in the local distribution of most reptiles made results difficult to interpret.

In general, reptiles were tolerant of fire-induced changes to their environment. Fire-related changes in population structure disappeared within a few years of fire and the reptile assemblage then appeared stable. This may have been due to the rapid regeneration and subsequent structural stability of *Banksia* woodland after fire.

INTRODUCTION

Australian ecosystems have developed under a long association with fire (Kemp 1981; Singh *et al.* 1981) and its significance to flora and fauna is widely-acknowledged. Fire is recognized as a valuable management tool and has been extensively used in management within the forestry industries of southern Australia (McCaw and Burrows 1989) and the pastoral industries of northern Australia (Lewis 1985). As noted by Friend (1993), however, its application in conservation has been restricted by a lack of understanding of its impacts, particularly upon fauna.

A number of studies, both in Australia and overseas, have found that diversity of plants and animals is highest soon after fire, with diversity declining in long-unburnt sites (Taylor 1973; Recher and Christensen 1981; Willan and Bigalke 1982). Willan and Bigalke (1982) also recorded a slight increase in small mammal diversity in long-unburnt sites. The general pattern of highest diversity soon after fire is associated with nutrient release and changes in the structural complexity of the vegetation as it regenerates and senesces after fire.

Detailed studies on vertebrate fauna in Australia have focussed mainly on mammals. In a detailed review, Friend (1993) concluded that small mammals have 'a reasonably consistent and predictable' seral response, with the response of a species to fire being consistent with its life history parameters such as shelter, food and breeding requirements. Similarly, Taylor (1991) concluded that with reference to small mammals, 'fire has been seen as a factor mainly influencing the availability of different successional stages of vegetation'. Numbers of most small mammal species decline dramatically after fire and individual species may achieve maximal abundance at a specific stage after fire when the vegetation satisfies their habitat requirements. This can give rise to successional-like changes in the levels of abundance of the species present (Fox and McKay 1981). Mammal species that have been most adversely affected by changes in fire regimes since European colonization are those that have the most specific habitat requirements in relation to post-fire stages of vegetation.

Observations on the impact of fire upon small mammals provide a theoretical framework for examining studies on other fauna, including reptiles. According to Friend (1993), reptiles are generally considered to be more resilient to the short-term impacts of fire than small mammals, but long-term changes in abundance have been recorded. Two studies (Cheal *et al.* 1979; Mather 1979, in Friend 1993) found that lizard species richness and abundance were lower in a long-unburnt than a recently-burnt site and associated this with understorey density and height. Caughley (1985) recorded little change in the lizard assemblage with time after fire but did find successional-like trends in relative abundances of the species. She associated this with changes in vegetation structure and composition, particularly in the case of the small agamid *Ctenophorus fordii*, which was most abundant soon after fire when the vegetation consisted of bare ground and dense clumps of regenerating plants. Kahn (1960) and Lillywhite and North (1974) made similar observations with the small iguanid *Sceloporus occidentalis*. Mushinsky (1985) examined the impact of fire frequency upon reptiles and found that some fire frequencies increased reptile density and diversity; he suggested this was due to the impact of regular fires on vegetation structure and particularly on the availability of open areas.

There appears to be some consistency between observations on the impact of fire on small mammals and reptiles. This has led Friend (1993) to suggest the development of a model to predict the response of a species to fire on the basis of its life history parameters. Available data on reptiles are limited, however, and this paper reports on a study of reptiles in relation to time since fire to see whether generalizations based on data collected on small mammals can be applied to this taxon.

METHODS

Study Area

The study area was near Mooliabeenee, approximately 80 km north of Perth at 31°20'S., 116°02'E. and consisted of six areas of Banksia woodland. These areas were similar in topography and the vegetation consisted of an understorey dominated by *Eremea pauciflora* and an overstorey to 6 m of *Banksia attenuata*, *Banksia menziesii* and scattered specimens of *Banksia ilicifolia* and *Eucalyptus todtiana*. The areas ranged in size from approximately 40 to over 100 ha but all were connected to larger regions of *Banksia* woodland; the layout of the areas is given in Bamford (1992a). Work was carried out on these areas from April 1983 to March 1986.

Sampling spanned the period 0-23 years after fire. Area 1 was burnt on 20 March 1985 and was previously burnt in 1962-63; it was thus sampled more than 20 years after and in the first year following a fire. Area 2 was burnt on 21 September 1984 and was sampled for a year following this fire to get some data on the impact of a low intensity, spring fire compared with all other fires, which occurred in summer-autumn and were of high intensity. Area 3 was burnt on 13 March 1983, 12 years after previously being burnt, and was sampled over the period 0-3 years after fire. Areas 4, 5 and 6 were burnt in March 1980, summer 1971-72 and summer 1962-63 respectively, and were thus sampled over the periods 3-6, 11-14 and 20-23 years after fire.

Sampling

Reptiles were studied using a mark-release-recapture program based on five consecutive nights' trapping each month with grids of 50 pitfall traps (hereafter referred to as pitfalls). Two such grids were placed in each area and within each grid, pitfalls were located at 5 intervals; each grid was therefore 45 m long by 20 m wide. The pitfalls consisted of 40 cm lengths of 15 cm diameter PVC pipe. They were capped when not in use and had 1.5 mm mesh bases to prevent specimens from tunnelling out.

In the first year of the study, only single grids were operated in areas 1, 4, 5 and 6, two grids being used in subsequent years and for all sampling in areas 2 and 3. Sampling on all grids was limited to two nights in January 1986 and no nights in February 1986. Therefore, expected captures on each grid for these months were calculated from the proportion of captures in previous years that occurred in January and February. Across all grids, captures in these months accounted for only 20 per cent of all captures. While numbers of captures could be estimated by this approach, numbers of species could not be similarly estimated. Therefore, analyses which required values for the number of species excluded estimated annual samples.

The experimental design used in this study was based on the opportunistic availability of sites; in the south-west of Western Australia it is very unusual to find a set of juxtaposed stands of vegetation varying in time since fire as did the sites used in this study. The sampling layout constituted pseudo-replication (*sensu*. Hurlbert 1984) because replicate grids were placed within the same site and the time-frame from 0-23 years after fire was spanned by a series of sites. This design places some restrictions on valid statistical analysis.

For analysis, the annual number of captures (including recaptures between months) of all species and of each species per grid were used as indices of abundance in relation to time since fire. As an index of abundance, the annual number of captures is almost certainly affected by differing trappability of species but was assumed to reflect differences in absolute abundance of the same species between sites.

Annual numbers of captures were analysed using a subdivided chi-square approach as described by Zar (1974). This involves testing a data set for significant variation, identifying the values which contribute most greatly to a significant result and then conducting another test with these values removed. If no significant variation is found in the modified data set from which outstanding values have been removed, values in the modified data set can be pooled and compared with each of the removed values to confirm the significance of the removed values. For this analysis, total annual captures per grid were treated as one data set, but annual captures per grid of all individual species except *Pogona minor* had to be pooled for each site as values were low, resulting in unacceptably small expected values.

Numbers of species and the Shannon-Weiner Diversity Index (H') were calculated for each annual sample. No analyses were conducted on these values, however, because both were affected by the large numbers of species represented by few individuals.

Samples of the same species from different areas were examined in detail to determine whether differences existed between them in population structure, as reflected by the distribution of snout-to-vent length (SVL) and the proportion of immature and adult specimens. From data on reptile morphometrics collected during the project, it was possible to estimate the size at sexual maturity for most species and, therefore, to recognize immature specimens on the basis of SVL and month of capture. All species bred in spring-summer and all except *Menetia greyii*, *Tympanocryptis adelaidensis* and possibly male *Morethia lineoocellata* reached sexual maturity at the end of their second year or later (Bamford 1986, 1992c). Where it was possible to compare proportions between areas or sets of areas, proportions were converted with the arcsine transformation before analysis (Zar 1974). Alternatively, mean SVL measurements were compared with Student's t-Test. In the study of population structure, *Varanus gouldii* and *Varanus tristes* were excluded as it was suspected that the pitfalls selectively trapped immature specimens of these species.

RESULTS

The Impact of Fire upon Levels of Abundance

Thirty species of reptiles were recorded during sampling (Table 1). An additional five species (the gekkonid *Underwoodisaurus milii*, the scincids *Tiliqua occipitalis* and *Morethia obscura* and the elapids *Pseudonaja nuchalis* and *Vermicella bertholdi*), were observed in the study region but were never trapped. The majority of species were trapped only in small numbers and therefore the impact of fire upon them could not be examined. Only eight species were represented by sufficient captures to allow species-specific analyses to take place.

Numbers of captures of all species, numbers of species and diversity (H'), based on annual samples in each grid, are presented in relation to fire in Figure 1. Annual numbers of captures varied significantly (χ^2 = 63.84, P<0.001), but this was due entirely to one annual sample of 72 individuals collected in area 3, 0-1 years after fire. There was no apparent pattern in species richness or diversity with time after fire.

Despite the lack of general trends in the reptile assemblage in relation to fire, numbers of captures of individual species were examined to see if trends were present (Fig. 2). Significant trends were found in *Pogona minor* (c^2 = 53.00, P<0.001), *Tympanocryptis adelaidensis* (c^2 = 101.24, P<0.001), *Lerista elegans* (c^2 = 99.33, P<0.001), *Lerista christinae* (c^2= 106.16, P<0.01), *Menetia greyii* (c^2 = 36.41, P<0.001) and *Morethia lineoocellata* (c^2= 116.70.33, P<0.001); but not in *Ctenotus lesueurii* (c^2 = 5.60) or *Cryptoblepharus plagiocephalus* (c^2=9.55).

Significant variation in captures of *Pogona minor* was lost (χ^2 = 29.95) with the removal of low captures in area 2 (0-1 years after a spring fire), and high captures in area 4 (4-5 years) and area 6 (20-21 years) from the data set. Comparison of each of these values with the pooled, modified data set confirmed their significance (c^2 = 6.28, P<0.025, c^2 = 9.47, P<0.005 and c^2 = 6.15, P<0.005 respectively). These results suggest that numbers may be low shortly after fire but high and variable with greater time after fire.

With *Tympanocryptis adelaidensis*, all significant variation was due to large numbers of captures in area 4 (3-6 years) (χ^2 =10.52 for the modified data set without captures from area 4). This was confirmed by the comparison of area 4 captures with the pooled, modified data set (χ^2 = 96.4, P<0.001). *Tympanocryptus adelaidensis* appears to be scarce after fire, abundant from 3-6 years after fire and then less abundant with greater time after fire.

Sources of significant variation were complex in *Lerista elegans*, with high captures in area 1 (0-1 and 20-22 years) and low captures in area 3 (0-3 years) and area 5 (11-14 years) being important (c^2 = 5.38 for the modified data set containing areas 2, 4 and 6 only). This wide source of significant variation suggests a patchiness in abundance independent of time since fire. Further analysis determined significant variation between the pooled, modified data set and area 1 (0-1 years) (c^2 = 39.3, P<0.001), area 1 (20-22 years) (c^2 = 23.6, P<0.001) and area 3 (0-3 years) (^{2} = 10.5, P<0.001), but not between the pooled, modified data set and area 5 (11-14 years) (c^2 =2.0). These results suggest that abundance may be greatest the longest time after fire, with high numbers of captures in area 1 (0-1 year) the result of individuals surviving after fire in the short term, although sampling in area 1 could not be continued to investigate this. The same results could have been achieved, however, if *L. elegans* was very patchily distributed, as areas 2 and 3 had no indication of *L. elegans* surviving from pre-fire abundance.

Variation in captures of *Lerista christinae* was due to large numbers caught in area 3 (0-3 years) and few captures in area 1 (0-1 years and 20-22 years) (χ^2 =8.41 for the modified data set without captures from areas 1 and 3). These results were confirmed by the comparison of the pooled, modified data set with area 3 (c^2 = 62.56, P<0.001) and area 1 (0-1 years c^2 = 3.92, P<0.05; 20-22 years, c^2 =5.85, P< 0.05). These results suggest that *L. christinae* is patchily abundant in *Banksia* woodland. There appeared to be a decline in

TABLE 1

Numbers of captures (including recaptures) of reptile species in all areas.

AREA	1	2	3	4	5	6	1
YEARS AFTER FIRE	0-1	0-1	0-3	3-6	11-13	20-23	20-22
TRAP NIGHTS	5900	7700	15 950	15950	12 450	12 350	7800
Gekkonidae							
Diplodactylus polyophthalmus		1	6				
Diplodactylus spinigerus	2		1	4	7	1	4
Pygopodidae							
Aprasia repens	1		6	3		1	2
Delma fraseri				1		1	1
Delma grayii	2						2
Lialis burtonis							2
Pletholax gracilis	2		3		9	2	
Pygopus lepidopodus			1				
Agamidae							
Pogona minor	11	8	39	51	40	64	28
Tympanocryptis adelaidensis		1	10	51	8	7	3
Varanidae							
Varanus gouldii		1	1				1
Varanus tristis			1	2		1	1
Scincidae							
Cryptoblepharus plagiocephalus	8	17	27	20	21	9	15
Ctenotus fallens	2	9	9	5	3	6	4
Ctenotus impar			1				
Ctenotus lesueurii	5	2	13	14	9	7	7
Ctenotus schomburgkii			1				
Egernia multiscutata					2		
Lerista christinae			80	16	17	16	1
Lerista elegans	20	5		6	2	11	21
Lerista praepedita	3	3			4	3	
Menetia greyii	12	30	16	11	17	8	9
Morethia lineoocellata			31	2	7	2	
Tiliqua rugosa				2	2		1
Typhlopidae							
Ramphotyphlops australis	2						
Elapidae							
Demansia psammophis			1				
Notechis curtus			1	1	1	3	3
Rhinoplocephalus gouldii			2		2	1	
Vermicella calonotus				1		1	
Vermicella semifasciata		1	1	1	3	2	1
N species	12	11	21	18	17	20	18
N captures	70	78	251	192	154	146	109

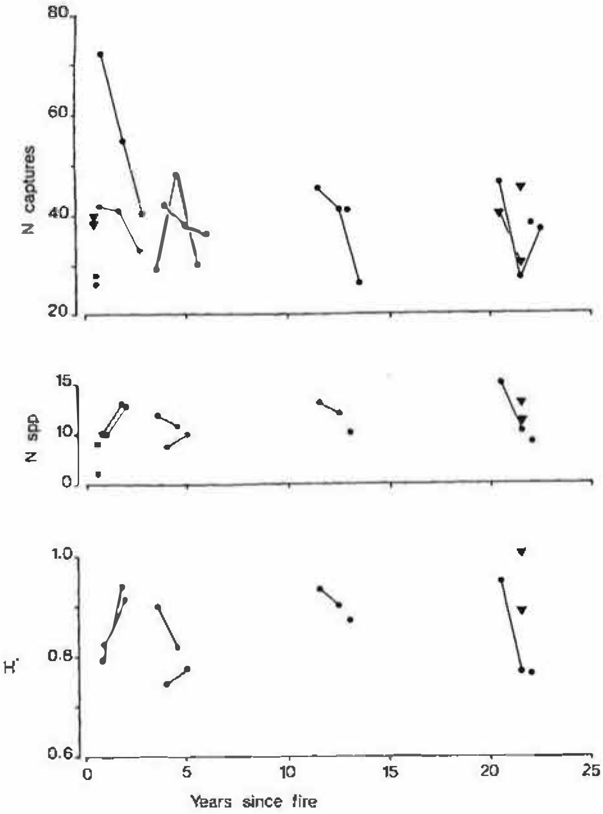

Figure 1. Number of captures (including recaptures) of reptiles per grid per year, number of reptile species per grid per year and Shannon-Wiener Diversity Index (H') of reptile captures per grid per year; against time since fire. Triangles represent data collected in area 1 before and after fire. Lines connect data collected on the same grid in different years. Number of species and diversity present data from complete years only, whereas number of captures includes values estimated from incomplete sampling in the third year of the study.

abundance in the first few years after fire (see Fig. 2); a sub-divided chi-square analysis of area 3 data only determined that significant variation was due to one annual sample only (c^2 = 18.9, $P< 0.005$ for all data; c^2 = 7.12 for modified data set excluding the highest annual capture).

Captures of *Menetia greyii* varied significantly as a result of samples collected in areas 1 (0-1 year) and 2 (0-1 year) (χ^2 =5.48 for the modified data set without captures from areas 1 and 3). In both these areas, significantly more than expected *M.greyii* were caught than in the pooled, modified data set (area 1 (0-1 year) χ^2 = 7.03, $P< 0.01$; area 2 (0-1 year) χ^2 = 29.58, $P< 0.001$). While these results suggest that the abundance of *M. greyii* may be higher in recently-burnt than long-unburnt areas, this observation was not supported by all sites.

Captures of *Morethia lineoocellata* were extremely variable and it was not possible to find a modified data set with no significant variation. Therefore, no patterns between significant variation and time since fire could be determined. This result could have been due to patchiness in the local distribution of the species.

The Impact of Fire upon Population Structure

Population structure in samples of *Pogona minor, Lerista christinae* and *Menetia greyii* are presented in Figure 3 and proportions of immature specimens are summarized for all species in Table 2.

Immature specimens dominated samples of *Pogona minor* collected more than three years after fire but were uncommon in some samples from more recently burnt areas. While area 1 (0-1 year) was an exception, almost all the immature specimens were caught within six weeks of the fire; these specimens had hatched before the fire. Without these immature specimens, area 1 would have had few captures after fire. In area 3, which was burnt at the same time of year as area 1, sampling did not begin until 8 weeks after fire and no immature specimens were caught in the first year. The cohort of immature specimens hatched immediately before an autumn fire appears to survive the fire but to disappear shortly after. Data from area 3 (1-2 years) suggest that breeding is limited or survival of immature specimens is poor in the second year after fire also. Data from area 2 (0-1 year, after a spring fire) suggest that the season (and possibly intensity) of fire is important to the survival of immature *P. minor*.

The proportion of immature specimens in samples of *Menetia greyii* was high in areas 3 (0-3 year) and 4 (3-6 years) compared with other areas, although numbers were too low to give a significant difference in the mean SVL (T = -1.01). Immature specimens hatched before the fire were caught in the autumn after fire in area 1 while immature specimens hatched before the fire in area 3 were not caught until the following spring when most had reached maturity; this reflected the 8-week delay in sampling in area 3. *Menetia greyii*, like other reptiles in the area, was rarely caught over winter. The sample from area 2 (0-1 years) was dominated by adult specimens and the significantly greater number of annual captures in area 2 compared with other areas was due to this. This large number of adults may have resulted from immigration, as *M. greyii* is believed to be nomadic with females concentrating in areas of low vegetation density to lay eggs (Bamford 1992b). Breeding was poor in the summer immediately following the spring fire, however, and no data were collected in the second summer after the fire.

In area 3, high captures of *Lerista christinae* were the result of large numbers of immature lizards, hatched before the fire, surviving in the first year after the fire. Immature specimens comprised only small proportions of samples with greater time after fire. Mean SVL was significantly less in area 3 (0-3 years) compared with all other areas (T = -3.82, $P<0.005$).

With the exception of *Tympanocryptis adelaidensis*, samples of most other species from recently-burnt areas tended to be dominated by immature specimens compared with samples from long-unburnt areas (Table 2). On the basis of mean SVL of samples with a large proportion of immature specimens being less than

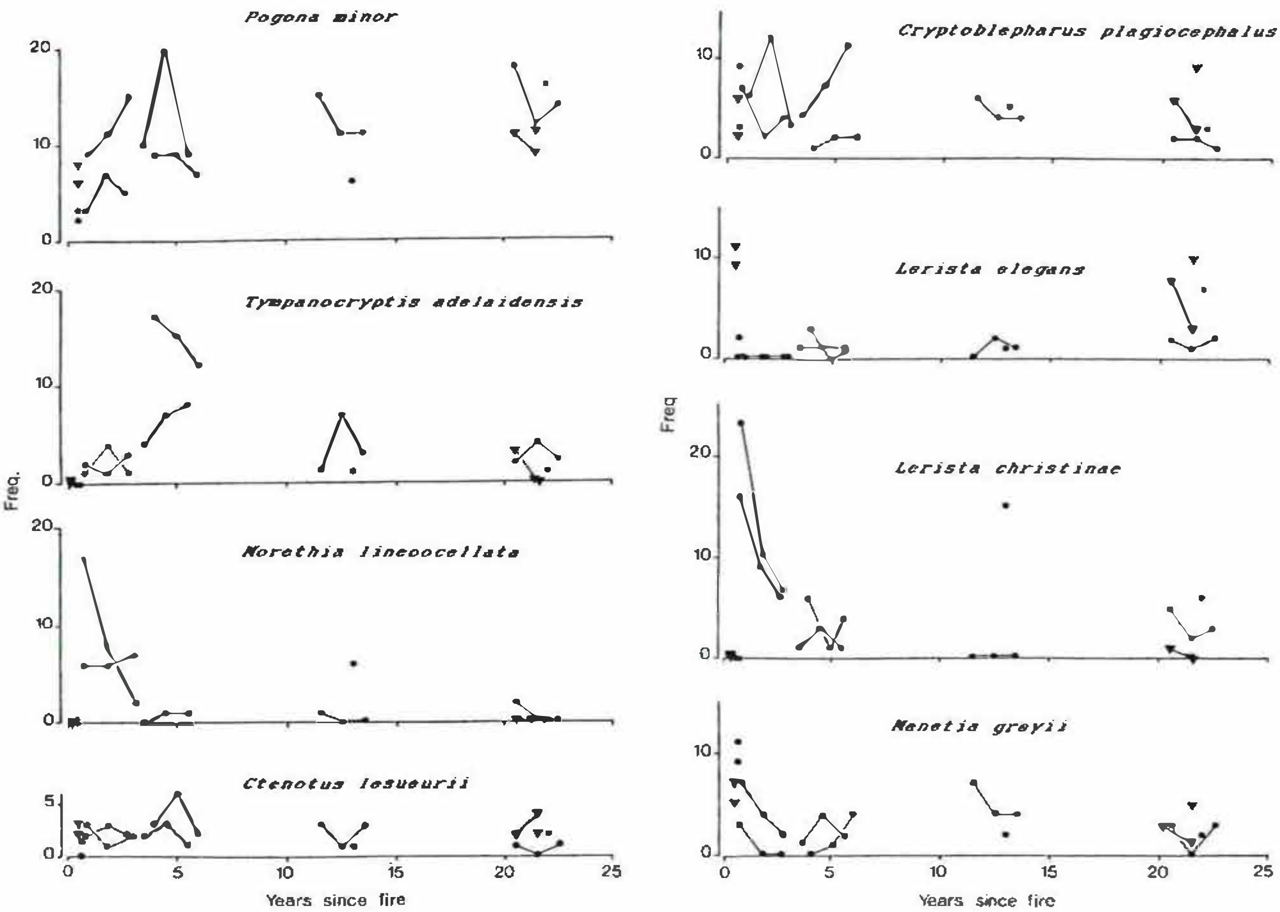

Figure 2. Numbers of captures of the most-often caught reptile species against time since fire. Triangles represent data collected in area 1 before and after fire. Lines connect data collected on the same grid in different years.

that of samples with fewer immature specimens, significant differences were found with *Cryptoblepharus plagiocephalus* ($T = -2.09$, $P<0.025$), *Morethia lineoocellata* ($T = -3.97$, $P<0.005$) and *Ctenotus fallens* ($T = -3.051$, $P<0.005$), but not with *Lerista elegans* ($T = -0.608$) and *Ctenotus lesueurii* ($T = -1.368$, although $P<0.10$). *Morthia lineoocellata* was distinctive in that the high proportion of immature specimens was recorded only in the second year after fire in area 3. *Ctenotus fallens* and *C. lesueurii* showed a similar tendency, but annual samples were too small to justify analysis. The representation of immature specimens in samples of *Cryptoblepharus plagiocephalus* differed from other species in that it was high only in area 2 (0-1 years, after a spring fire). The number of specimens involved was small but four of the five immature lizards recorded hatched in the summer after the fire while one was a survivor from the previous breeding season.

Pogona minor was the only frequently-caught species which showed a decline in the proportion of immature specimens in the first year after fire. Even those species caught too infrequently to examine individually, when pooled, revealed a trend for a greater proportion of immature specimens in recently burnt areas (Table 2). For all species except *P. minor* pooled (Table 2), the mean proportion of immature specimens from samples collected over the period 0-6 years after fire was significantly greater than for samples collected over the period 11-23 years after fire ($T = 4.18$, $P<0.005$).

DISCUSSION

In studies of relationships between reptile assemblages and time since fire, a number of patterns have been observed. These include: greater richness and abundance in recently-burnt compared with long-unburnt sites (Cheal *et al.* 1979; Mather 1979, in Friend 1993) and successional-like changes in relative abundance with increasing time after fire (Caughley 1985). None of these was recorded in the Mooliabeenee study.

Impacts that could be related to fire were generally restricted to the first few years and in most cases consisted of short-term changes in population structure. Three of the six species affected showed increases in numbers of captures in the first year or few years after fire, one showed an increase in the period 3-6 years after fire, one showed an increase 20-23 years after fire and one showed a decline in the first year after

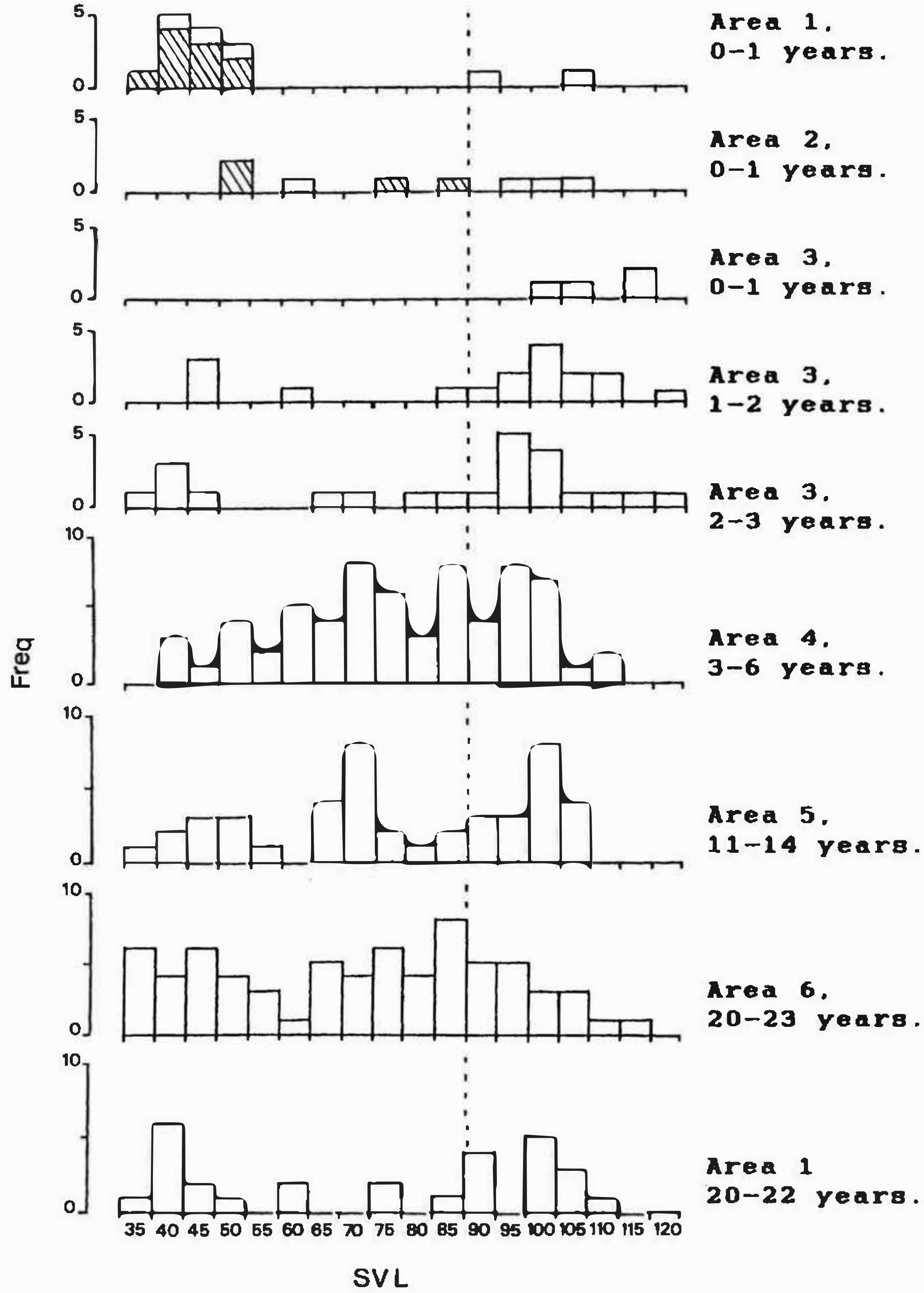

Figure 3 (a). Distribution of snout-to-vent length in samples of Pogona minor, *in relation to time since fire. Only the lower value of each snout-to-vent length category is indicated. Hatching indicates immature specimens hatched before and area was burnt and caught in the first year after fire. The broken vertical line indicates SVL at sexual maturity. Note that sampling effort was not the same for each histogram; sampling efforts are given in Table 1.*

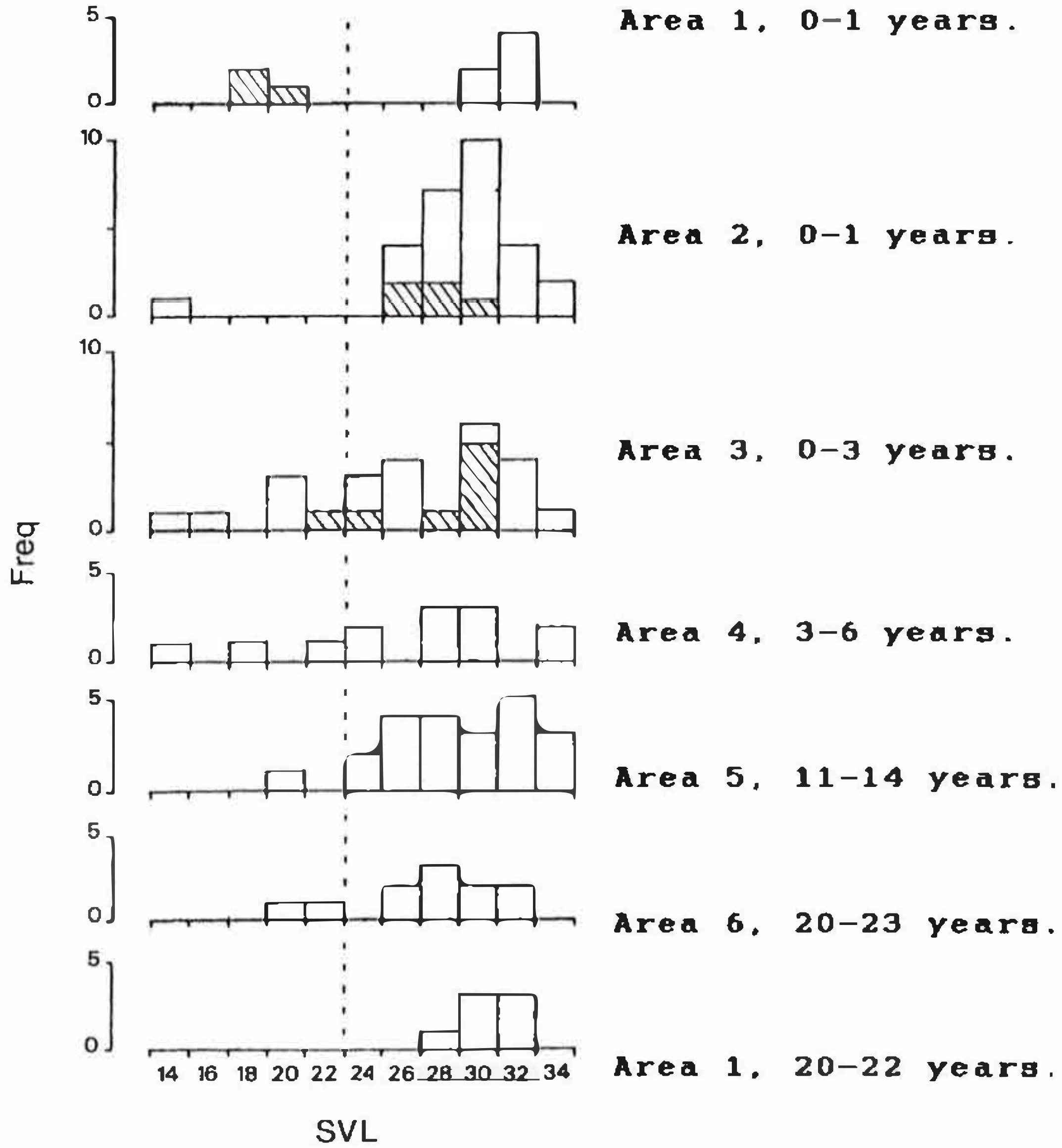

Figure 3 (b). Distribution of snout-to-vent length in samples of Menetia greyii *in relation to time since fire. Only the lower value of each snout-to-vent length category is indicated. Hatching indicates immature specimens hatched before and area was burnt and caught in the first year after fire. Note that sampling effort was not the same for each histogram; sampling efforts are given in Table 1.*

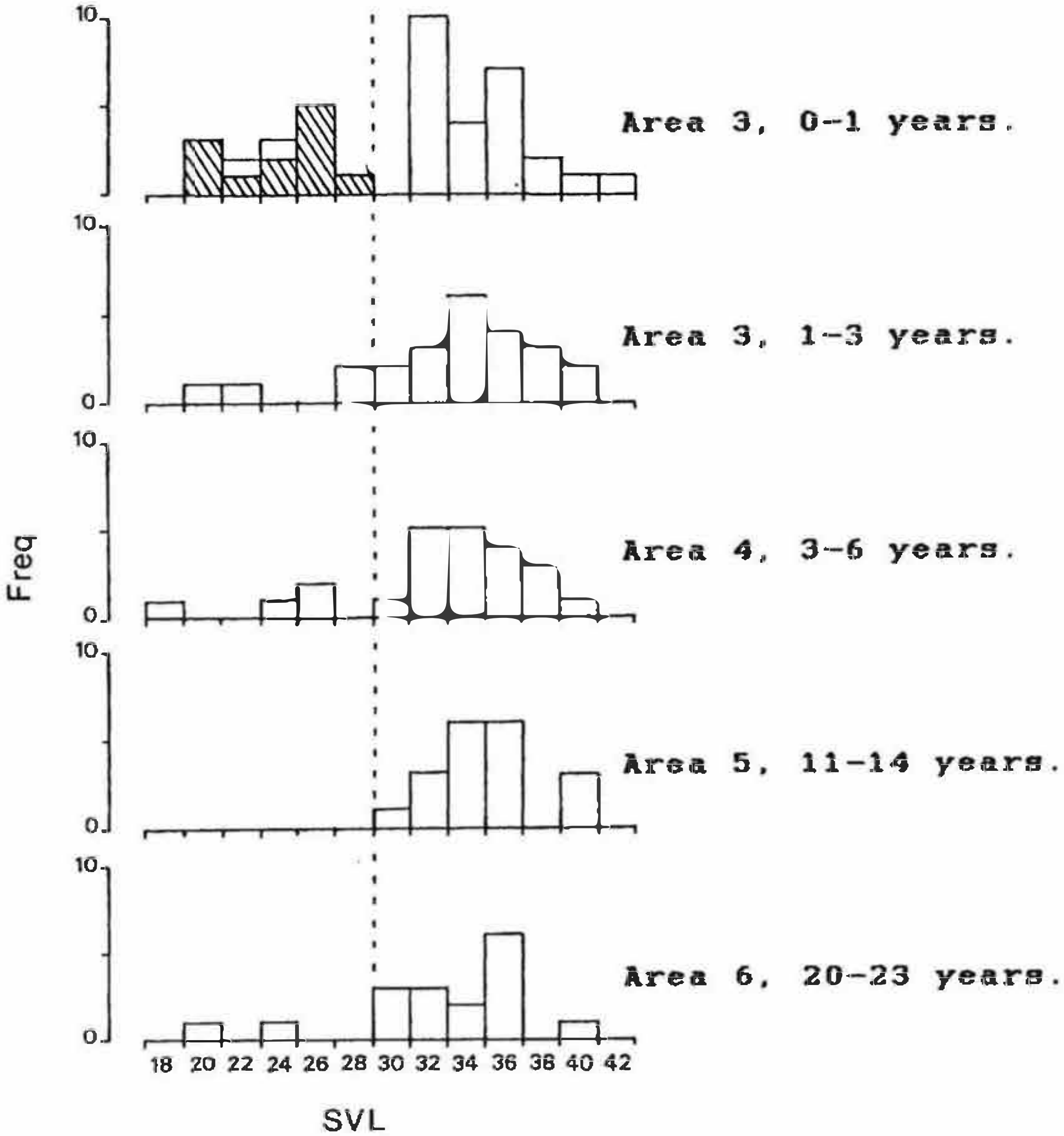

Figure 3 (c). Distribution of snout-to-vent length in samples of Lerista christinae *in relation to time since fire. Only the lower value of each snout-to-vent length category is indicated. Hatching indicates immature specimens hatched before an area was burnt and caught in the first year after fire. Note that sampling effort was not the same for each histogram; sampling efforts are given in Table 1.*

TABLE 2

Proportions of immature reptiles in samples in relation to time since fire. Data from areas of similar time since fire pooled to generate adequate sample sizes.

SPECIES	AREAS	YEARS SINCE FIRE	SAMPLE SIZE	PERCENTAGE IMMATURE
Pogona minor	1,2,3	0-3	47	48.9
	4	3-6	56	57.1
	5	11-14	40	55.0
	1, 6	20-23	94	64.9
Tympanocryptis adelaidensis	3	0-3	12	25.0
	4	3-6	52	26.0
	1,5,6	11-23	13	23.1
Cryptoblepharus plagiocephalus	2	0-1	17	33.0
	1,3	0-1	35	5.7
	4	3-6	20	15.0
	1,5,6	11-23	45	6.7
Lerista christinae	3	0-1	32	31.2
	3	1-3	24	16.7
	4	3-6	16	12.5
	1,5,6	11-23	31	0
Lerista elegans	1	0-1	17	29.4
	1,5,6	11-23	32	12.5
Menetia greyii	1,2,3	0-1	45	13.3
	3,4	1-6	15	20.[illegible]
	1,5,6	11-23	32	6.2
Morethia lineoocellata	3	[illegible]-1	14	0
	3	1-3	22	50.0
	4,5,6	3-23	12	8.3
Ctenotus fallens	1,2,3	0-3	20	85.0
	1,4,5,6	3-23	16	31.2
Ctenotus lesueurii	1,2,3	0-3	20	70.0
	1,4,5,6	3-23	37	43.2
Other species	1,2,3	0-1	25	36.0
	3,4	1-6	30	6.7
	5	11-14	32	15.6
	1,6	20-23	34	8.8
all species except *P. minor*	1	0-1	55	38.2
	2	0-1	65	33.8
	3	0-1	86	22.1
	3	1-3	105	31.4
	4	3-6	138	23.9
	5	11-14	111	10.8
	6	20-23	80	13.8
	1	20-22	73	9.6

fire. In four of these species, however, patchiness in their local distribution complicated the results.

There was no indication, from changes in the frequency of recaptures or distances moved by recaptures, that observed impacts were due to changes in behaviour rather than to population changes. It is also unlikely that specimens identified as immature were actually stunted adults, as they were of the expected size at the expected time of year. The only exception was a single *Pogona minor* in area 2, caught when about 21 months old and therefore at the beginning of its first breeding season, but noticeably smaller than specimens of similar age elsewhere. The spring fire occurred when this specimen was about nine months old and may have inhibited its growth over the following summer.

Several different patterns in the short-term changes in population structure emerged. In *P. minor*, a decline in abundance after fire was linked to a decline in the abundance of immature lizards in the first six months after an autumn fire. *Pogona minor* is partly arboreal and the lack of understorey cover may have led to increased predation upon small specimens. A more strictly arboreal species, the gecko (*Diplodactylus spinigerus*), was infrequently caught but appeared to suffer a similar post-fire decline. Adult *D. spinigerus* are about the same size as recently-hatched *P. minor* and were very abundant in the first week after fire in area 1. Normally inconspicuous, their pale grey coloration made them very obvious as they clung to blackened twigs. They had survived the fire by sheltering in burrows of other species, including those of the large scorpion *Urodacus novaehollandiae* (personal observation).

The small, fossorial *Lerista christinae* was very abundant in area 3 after fire owing to large numbers of immature specimens from the breeding season before the fire. Survival of this cohort may have increased owing to the complete absence in the first year after fire of the predatory dasyurid marsupial *Sminthopsis griseoventer* (Bamford 1986).

Patchiness in local distribution in this and other species, such as *Lerista elegans*, may have concealed an increase in abundance with greater time after fire. Such an increase could have been associated with increased leaf-litter and would support the relationship between habitat selection and response to fire, but the data are inconclusive.

In *Morethia lineoocellata* and possibly also *Menetia greyii*, *Ctenotus fallens* and *C. lesueurii*, high proportions of immature lizards in the first few years after fire were due to enhanced recruitment in the first breeding season after fire, not to enhanced survival of young from the season before the fire as seen with *L. christinae*. *Cryptoblepharus plagiocephalus* displayed a similar pattern, but only after a spring fire. Such enhanced recruitment might be expected to result in an increase in abundance levels in the reptile assemblage, but appeared to be too short-lived an effect to have such an impact. Enhanced recruitment may have led to the significantly high levels of abundance of *Tymparocryptis adelaidensis* 3-6 years after fire, however. Enhanced recruitment may have gone undetected in this species because individuals mature in less than one year and rarely live for more than two (Bamford 1992c).

Tymparocryptis adelaidensis was the only species with a response to fire that could be interpreted using the hypothesis that a species' habitat preferences influence the impact of fire upon it. Bamford and Bamford (1992) found that captures of *T. adelaidensis* along two, 3 km transects of pitfalls, established through *Banksia* woodland without initial reference to vegetation structure, were concentrated in pitfalls associated with low densities of vegetation. This was at a site 100 km north of Mooliabeenee in an area unburnt for about 12 years. It was suggested that the species actually favoured locations where dense vegetation and open areas were juxtaposed. Such patchiness in vegetation density was evident 3-6 years after fire.

The response of *T. adelaidensis* to fire, involving a peak in abundance a few years after fire associated with patchiness in vegetation density, is similar to that observed in *Ctenophorus fordii* (Caughley 1985) and *Sceloporus occidentalis* (Kahn 1969; Lillywhite and North 1974). Mushinsky (1985) also stressed the importance of patchiness in vegetation density produced by some fire regimes for reptiles. *Tymparocryptis adelaidensis*, *C. fordii* and *S. occidentalis* are small representatives of two families often thought of as ecological analogues (Pianka 1971). A preference for patchy vegetation would appear to be responsible for their similar responses to fire.

Long-term changes with fire found with other studies upon reptiles and very evident with studies on mammals (Friend 1993), have been related to habitat preferences and changes in vegetation structure after fire. In the present study, however, only the response of *T. adelaidensis* to fire can be explained in terms of habitat selection. In other species for which sufficient data were collected, the response was brief and related to changes in survival of immature lizards. The absence of long-term responses in the present study may reflect the speed of regeneration of *Banksia* woodland after fire. Both understorey and overstorey plants resprout after fire and changes to vegetation structure over the period about 5-23 years after fire are minor. This brief period of rapid regeneration is within the lifespan of most of the reptiles recorded in the area (Bamford 1986). In this respect, the fast-maturing, short-lived *T. adelaidensis* is unusual and resembles *C. fordii* (Heatwole 1976). With low rates of population turnover, the populations of most reptiles may not be greatly affected by one or two years of altered recruitment after a fire, while those species with predictable responses to fire may tend to be fast-maturing and short-lived.

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FIRE-INDUCED LANDSCAPE MOSAICS

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Fire intensity and the maintenance of habitat heterogeneity in a tropical savanna

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CALMScience Supplement 4: 189-196 (1995)

Fire intensity and the maintenance of habitat heterogeneity in a tropical savanna

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ABSTRACT

Savanna vegetation is associated with a highly seasonal tropical climate which results in high-frequency grass fires. Other studies have shown that there are strong relationships between habitat heterogeneity or patchiness and mammal richness and abundance in the savannas of Kakadu National Park. Fires vary greatly in intensity and impact on vegetation. Both pronounced patchiness and differentiation of ground and canopy strata are fundamental characteristics of savannas. A model to account for what appears to be a dynamic stasis is presented. In this model, low intensity fires create great patchiness in the ground layer. In contrast, both high-intensity fires and absence of fire create patchiness in the tree layer, owing to differential tree mortality and recruitment from the ground layer respectively.

Experimental fires in Kakadu showed different effects on the tree and ground layers, lending some support to the above model. Fire-induced changes in vegetation patchiness, dominant plant species richness and mammal abundance and richness were apparent. It is likely that a fire regime approximating the pattern created by traditional Aboriginal burning is the best management for savanna biodiversity.

INTRODUCTION

The pattern of any vegetation changes with time. The speed at which this happens is determined by the frequency and intensity of disturbance and also the productivity of the system. Huston (1979) has argued that maximum species diversity occurs when the frequency/intensity of disturbance maintains the community in a constant state of disequilibrium. The species never get to a stage where competitive exclusion would occur and thus loss of species is avoided. Denslow (1980) has, in turn, argued that the maximum species diversity is obtained with disturbances which are of the historically most common size. Frequency and intensity of disturbance, the size of the disturbance patch, and the life history of the species affected are all correlated (Braithwaite and Estbergs 1985).

Regions of greater habitat diversity (for topographic/geological reasons) are often richer in species of animals (i.e. alpha diversity; Woinarski and Braithwaite 1990). Whether habitat diversity can be manipulated with fire to produce increased animal diversity has never, to my knowledge, been tested.

Vegetation patchiness or habitat heterogeneity seems important for a number of reasons.

1. It has become recognized as one of the three main components of biodiversity, the others being genetic diversity and species diversity (e.g. Groombridge 1992).
2. It may be an important predictor of species diversity and animal abundance (e.g. Woinarski and Braithwaite 1990). The scale of the patchiness may be what is most critical.
3. It may represent an appropriate goal for manipulating habitat for maintaining vulnerable mammal species (Braithwaite 1985a).
4. It offers the possibility of identifying areas of high biodiversity using remote sensing.

Savanna ecologists have long identified the four major determinants of tropical savannas as plant available moisture (PAM), available nutrients (AN), fire and herbivory (Frost *et al.* 1986). The first two are regarded as the primary determinants and the latter two are secondary modifiers of savannas. Fire enables eucalypt savanna to push back the boundaries of rainforest or monsoon forest patches (Bowman and Dunlop 1986) and can eliminate them entirely (Bowman and Wightman 1985). Conversely, in the absence of fire, rainforest, monsoon forest and other fire-sensitive species can slowly invade the savanna (Bowman and Fensham 1991) and at least give rise to a more shrubby understorey (Hoare *et al.* 1980).

In the literature on tropical savannas, the tree or canopy layer and the ground vegetation layer have been seen as semi-independent, competing but co-existing systems (Walter 1971). There have been studies of competition between the root systems of the two strata, with the canopy roots predominating in deeper soil horizons (Knoop and Walker 1985). Braithwaite and Estbergs (1985) have shown that the canopy experiences fires much less than the ground layer. The species of the canopy are long-lived compared with those of the ground layer which is dominated by annuals, short-lived perennials, and the long-persisting juveniles of the canopy species.

In Australia particularly, the savanna seems remarkably stable under what appears to be a wide variety of fire regimes. It is a paradox: how can the savanna be burned and change in so many different ways yet remain savanna ? How does the patchiness of the savanna relate to the species diversity of the biota? A simple patch dynamic mechanism for this is proposed and preliminary data from Kakadu National Park examined in its light.

A MODEL OF SAVANNA HOMEOSTASIS

While climatic and edaphic factors are the major determinants of the relative dominance of canopy and ground layers in savanna, the balance of dominance can move from one stratum to the other, with fire and herbivores a common agent of change (Belsky 1990). In Australia, fire with some facilitation from wood-eating termites, is the main cause of tree loss (*c.* 70 per cent of total), with windthrow (*c.* 10 per cent) and lightning (*c.* 10 per cent) and termites alone (*c.* 10 per cent) as minor agents (Braithwaite 1985b), and absence of fire the main facilitator of recruitment to the canopy (Hoare *et al.* 1980). The concept of 'clump-interclump' (Hoare *et al.* 1980) or fine scale patchiness of domination by trees or grass, is one of the defining elements of savanna. However, the patchiness within each of the ground and canopy layers is also a typical savanna characteristic which both determines and is determined by fire (Stott 1986).

It is well known, at least in tropical Australia, that early dry season fires are patchy on the ground (Jones 1980). These fires typically burn only part of the ground cover within an area of ignited savanna (Kapalga sub-catchments mean=73.8 per cent, range= 30-99 per cent, n=9), leaving many individual small plants untouched by fire (Braithwaite and Estbergs 1985). However, as the young plants are still actively growing at this time of year, they are vulnerable and experience greater mortality during early dry season fires than in the typically more intense late dry season fires (P.A. Werner, personal communication, 1990). Thus I hypothesise that it is these early dry season fires which cause greatest patchiness in ground layer vegetation composition. On the other hand, at very low intensity or no fire, the existing plants grow in size, but little change in patchiness occurs. However, species may be gradually lost through competitive exclusion (*cf.* Huston 1979), or richness increase due to absence of adversity or increase in favourableness, depending on the type of patch. Similarly, at high intensity, the ground cover receives a much more homogeneous treatment by fire, diminishing patchiness. Patchiness of the ground layer is thus also consistent with the intermediate disturbance hypothesis (Connell 1978).

The situation is different with the canopy. Typically, early dry season fires impact minimally on the canopy (Braithwaite and Estbergs 1985). In fact, the lack of impact on the flowering of fruit trees was a stated reason for the concentration of Aboriginal burning at this time (Haynes 1985). However, high intensity late dry season fires typically scorch high into the canopy (Braithwaite and Estbergs 1985) and this occurs during the period of leaf-flush before the wet season (Braithwaite 1985). Such fires are likely to cause maximum mortality (Lonsdale and Braithwaite 1991) and may create opportunities for subsequent recruitment into the canopy (Fensham and Bowman 1992). With no fires or fires of low intensity, major change in the canopy occurs due to the recruitment of plants out of the ground layer (Hoare *et al.* 1980).

The pattern of change described in the preceding paragraphs is summarized in Figure 1. Change in patchiness can be positive or negative depending on the loss or gain of individuals of fire-sensitive plant species. The degree of change in relation to fire intensity in the canopy layer is opposite to that for the ground layer. If the intensity is low, relative importance shifts towards the canopy and if it is high, it shifts towards the ground layer. At intermediate intensities, transition between the two is minimal. In combination the elements of this model can maintain savanna as a dynamic stasis. While soil moisture and nutrients are the primary determinants of vegetation type, the key to whether savanna changes to grassland or rainforest or remains as savanna is often the ambient fire regime (Frost *et al.* 1986).

In rainforest studies, Denslow (1980) has argued that the most common patch size will support the highest diversity of species. The historic disturbance regime will produce patch sizes which have been most common historically. This is one reason why the documentation of the traditional fire regime of indigenous hunter-gatherers is of significance for contemporary conservation. However, as Braithwaite (1992) has argued, the traditional regime as expressed on the landscape as a whole, is an emergent property of the activities of many quasi-independent individuals and groups. It was not the result of a brilliant ecological master-plan which we can simply take down from the historical shelf and naively apply in a contemporary context (Redford 1991) of roads, tourists, exotic species problems, contemporary technology, and changed Aboriginal culture. It is necessary to reconstruct the historic pattern of burning and adapt it as best we can using knowledge from any source.

Braithwaite (1991) described the seasonal pattern of Aboriginal burning using the historical record from the nineteenth century and found it corresponded well with the results of independent ethnographic studies. The traditional Aboriginal regime had a peak of burning in the early dry season. However, some fires did occur at any time during the 9.5 month fire season (Braithwaite 1991). It has been found by National Park managers that the more country that is burnt during the early dry season the less that is burnt during the late dry season (Press 1988). This is partly due to the fragmentation of the savanna fuel induced by the early fires. The late fires do not carry as far because of the lack of continuity of fuel. For the same reason, parcels of land remain unburnt in an area which had received heavy early burning. Thus the three important elements necessary for the model of savanna homeostasis to work were integral to the traditional Aboriginal burning regime, early fires with some late fires and some areas unburnt.

It follows from Denslow (1980) that the traditional regime described above would maximize species diversity. Thus the proposed model would predict the persistence of the diversity of savanna vegetation. The savannas of north-western Australia have a distinctive and diverse mammal fauna richer than the local rainforest (Braithwaite *et al.* 1985). Thus it might be expected that at least the mammal fauna of the Australian savanna would also operate in synchrony with the patch dynamics of the savanna vegetation.

The direction of changes resulting from the fires of different intensity is likely to be both positive and negative. Because all community change is the result of natality, survival, migration and mortality of individuals of different species, the direction of change is often not predictable. The increase (or decrease) in abundance of a given species may increase or decrease patchiness and may increase or decrease the attractiveness of a piece of land for mammals and other animals.

In this paper, the above model is tested with experimental data on vegetation patchiness, plant and mammal species richness.

METHODS

Within Kapalga Research Station in Kakadu National Park (12° 43' S, 132° 26' E), pairs of 8-ha trapping grids (totalling 128 ha) were established, one located in riparian/woodland vegetation and the other about 0.5 km upslope in open forest. Thus the grids spanned an elevational (20 m) and moisture gradient from riparian vegetation through woodland and into the locally dominant *Eucalyptus miniata- Eucalyptus tetrodonta* open forest association. *Sorghum* spp. was the dominant grass understorey (see also Andersen and Braithwaite 1994). Pairs of grids were located in eight different 15-20 km^2 sub-catchments in the southern half of Kapalga Research Station. All sites occurred within a 300 km^2 area. Each of the 16 grids was arranged as 4 rows at 50 m spacing with 20 trapsites at 20 m spacing. They were trapped with 1:4 (wire:Elliott) trap type ratio for two nights every two months. The trapping study went from July 1989 to July 1992.

Twelve species of mammals (Appendix 1), ranging in size from *Pseudomys delicatulus* (10 g) to *Trichosurus vulpecula* (2 kg), were trapped for a total of 46 080 trap nights during the study. The species were classified (Appendix 1) into arboreal (spends some time in trees) and terrestrial (spends no time in trees).

Following pre-fire trapping from July 1989 to May 1990, experimental fires were set in 1990 and 1991 as part of the landscape-scale Kapalga Fire Experiment (Braithwaite 1990; Williams 1994). Two catchments (four grids) were burned in June of both years (Early), two in both Septembers (Late), and two in June in 1991 and 1992 with some remaining unburnt areas also burnt in July and September in 1992 (Progressive). The Progressive regime was an attempt to simulate the Traditional Aboriginal habit of progressively burning later-drying parts of the landscape. At this early stage in the experiment, Progressive as a treatment was not substantially different from Early and has been treated as single annual fire. Another two catchments (four grids) were unburnt throughout. This regime was called Natural as it was initially unclear whether they would be burnt by lightning-ignited fires.

The division of the vegetation into canopy and ground strata at 3 m was based on previous analyses of vegetation structure (Braithwaite and Estbergs 1985). Three measures of patchiness were used. Measures of the patchiness of the fires, of the ground cover composition (<3 m) and the canopy composition (>3 m) were derived.

The patchiness of the fires was measured using the coefficient of variation of scorch height/total height. After each fire the height of scorching (maximum height at which most leaves were dead) and total canopy height were estimated at each trap site. Any scorch height measure is partly confounded by variation in canopy height between sites but the ratio was thought to give more reliable estimates in the present context of minor canopy height variation.

For each of the 16 trapping grids, a total of 80 estimates was made. The means and coefficients of variation were used. Estimates of intensity in kW.m^{-1} (see Williams 1994) from the vicinity of the grids can be derived from the mean scorch height/height ratios (per cent) for 1991 and 1992 fires ($y=3.301+ 87.86x$, $r=0.674$, $p=0.02$). The fire intensities ranged from about 3000 to 12 000 kW.m^{-1}.

The patchiness of the ground and canopy vegetation was estimated separately. The dominant vegetation on each of the 1280 trapping sites was measured in February of each year. The five species with the highest cover/abundance for above 3 m and for below 3 m were recorded with the Braun-Blanquet estimate. For vegetation <3 m, a 1 m quadrat was used, while for >3 m a 10 x 10 m quadrat was used. The sampling was first done in February 1990 before the first fires in May 1990. The sites were previously burnt by intense late fires in 1985 and 1986, low intensity early fires in 1987 and no fires in 1988 and 1989. Subsequent sampling was done in February 1992 for >3 m and in February 1991 and 1992 for <3 m.

The 1280 sites from different years were classified in <3 m and >3 m data sets using the non-hierarchical, agglomerative clustering program ALOC (Belbin 1987). The >3 m data were for two years (1990 and 1992, 2560 sites) and yielded 25 vegetation groups at the 0.99 heterogeneity level. The <3 m data were analysed in two ways. First, the three years (1990-1992; 3840 sites) were clustered into 15 groups at 0.99 heterogeneity level. Second, the 1991 and 1992, the first two post-fire years were clustered into 27 groups at 0.99 heterogeneity level. Measures of patchiness were derived by calculating a Shannon-Weiner diversity statistic using the numbers of sites of each vegetation group on a trapping grid.

RESULTS

Change in Vegetation Patchiness

Vegetation diversity or patchiness in vegetation composition changed little between 1990 and 1992 for the >3 m fraction but changed greatly between years for the <3 m fraction. For example, the correlation coefficients between years were high when there was little change as in >3 m ($r=0.935$, $p<0.001$) but low for <3 m (1990-1 $r=0.601$, $p<0.05$; 1991-2 $r=0.681$, $p<0.01$; 1990-2 $r=0.705$, $p<0.01$), when change was much greater.

Measures of change have been plotted against mean scorch height/total height percentage (scorch/height, Figs 2-4). It should be remembered, however, that the coefficient of variation of scorch/height, a measure of the patchiness of the fires themselves, increases from zero for no fires to reach highest values between 20 and 60 per cent and decreases steeply as maximum scorch height is approached.

Change in vegetation patchiness from pre-fire to post-fire is plotted against mean scorch height/total height for the 16 trapping grids (Fig. 2). As an aid to interpretation, lines have been drawn in by eye indicating the possible shape of the envelope containing the cloud of data points. The precise shape of these lines is not important for the model presented. The important thing for testing the model is consistent difference between the ground and canopy layers (see Fig.1).

Substantial change in vegetation patchiness took place for vegetation <3 m from 1991 to 1992 (first to second post-fire year) (Fig. 2a). The change is both positive and negative but most is negative (decreasing patchiness). The most strongly positive changes are the Progressive sites on compartment A (see Williams 1994). The Natural sites with no fires are of course at zero scorch/height. They form a tight group with a slight decrease in patchiness. The Late treatment sites at the other end of the intensity scale show tighter grouping than the other two remaining groupings. The Early and Progressive sites show greatest change but also greatest variability between sites. The overall pattern is one of larger change at the intermediate levels of fire intensity (scorch/height) and less change at high and low intensity.

When the same approach is taken for the canopy (>3 m) vegetation, a different result is obtained (Fig. 2b). The degree of change is much less than for the ground layer. The change in canopy patchiness is greatest at high and low mean fire intensities. The intermediate level of intensity results in least change in canopy patchiness. The no or low intensity situation results in change in patchiness consistent with recruitment from the ground layer (i.e. some of the small individuals are able to grow sufficiently to get into the canopy). The effect of fire protection on the canopy patchiness of the Natural sites was still being played out after five years fire protection. At the other extreme, the very intense fires increase patchiness consistent with substantial mortality of some species of trees and shrubs.

Change in Plant Richness

Because only the dominant five plant species for each of <3 m and >3 m categories were recorded, a complete vascular plant inventory is not available for the sites. The dominant species richness value is the total number of species recorded on an 80-site trapping grid. Change in species richness is plotted against scorch/height for <3 m in Figure 3a. The patterns are for 1991-92 and show an increase in Natural sites and a decrease in the more intensely burnt Late and Early sites. This produces a negative relationship between mean scorch/height and change in plant richness ($r=-0.523$, $p=0.037$). The pattern is also similar to that obtained for vegetation patchiness in that greatest variation in change in plant richness occurs at intermediate fire intensities (around 60 per cent scorch/height).

Above 3 m, the pattern obtained (Fig.3b) is also similar to that with vegetation patchiness. That is, the intermediate fire intensities (30 per cent) show least variation in degree of change. The increase in richness on the unburnt sites and the decrease in plant richness on the high scorch/height sites confirms the interpretation of the change in vegetation patchiness described above ($r=-0.578$, $p=0.019$). That is, above 3 m, the change is consistent with recruitment of new species at the low intensity end and by loss of species through mortality at the high intensity end.

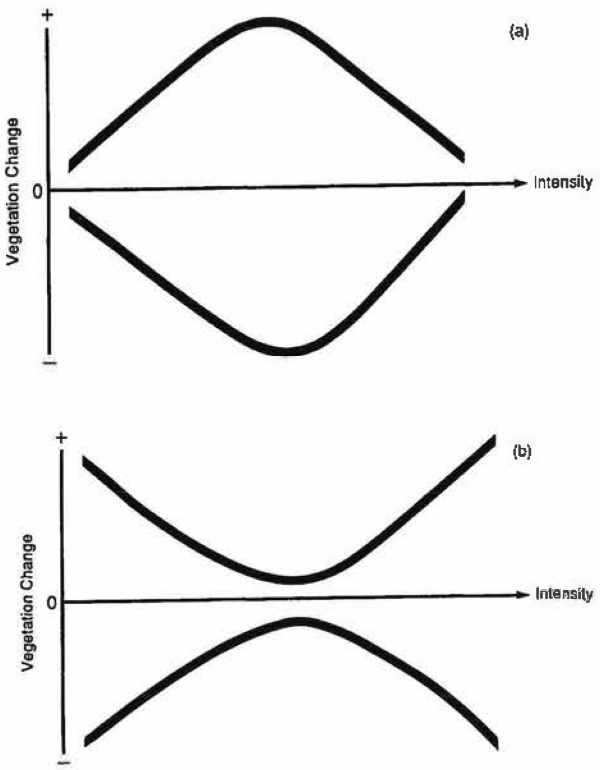

Figure 1. A model of change in habitat characteristics (habitat diversity and plant and animal species richness) in relationship to fire intensity for ground (a) and canopy (b) storey savanna vegetation.

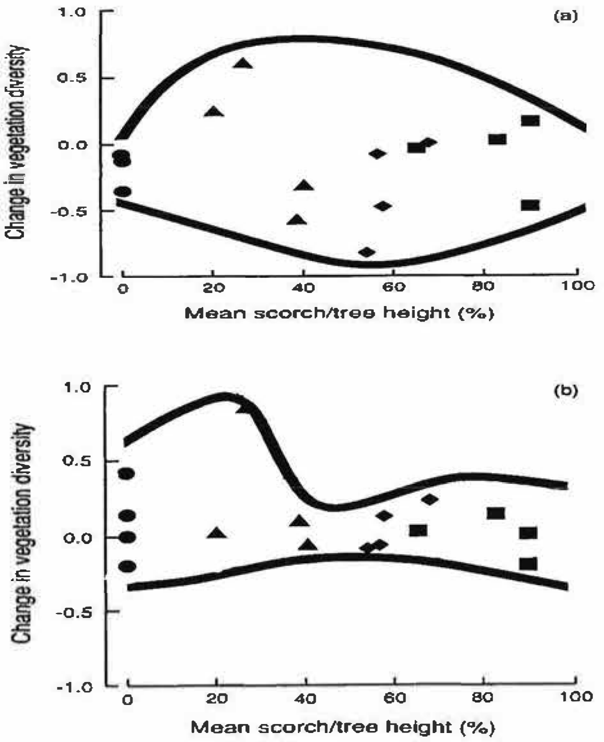

Figure 2. Change in vegetation diversity in relation mean scorch height /total height ratio for 1990 and 1991 for (a) ground and (b) canopy layer savanna vegetation. The fire regimes were Natural (circles), Early (diamonds), Late (squares), and Progressive (triangles).

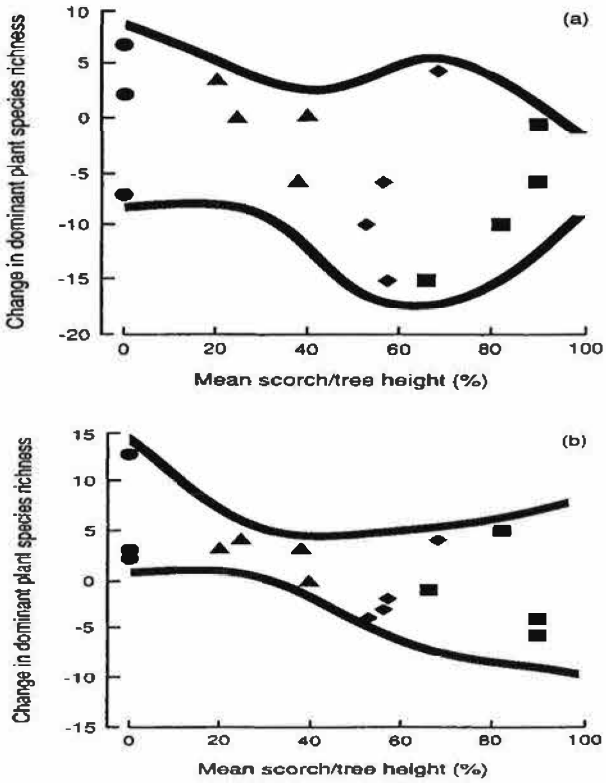

Figure 3. Change in dominant plant species richness in relation to mean scorch height/tree height ratio for (a) ground and (b) canopy layer savanna vegetation. Symbols as in Fig.2.

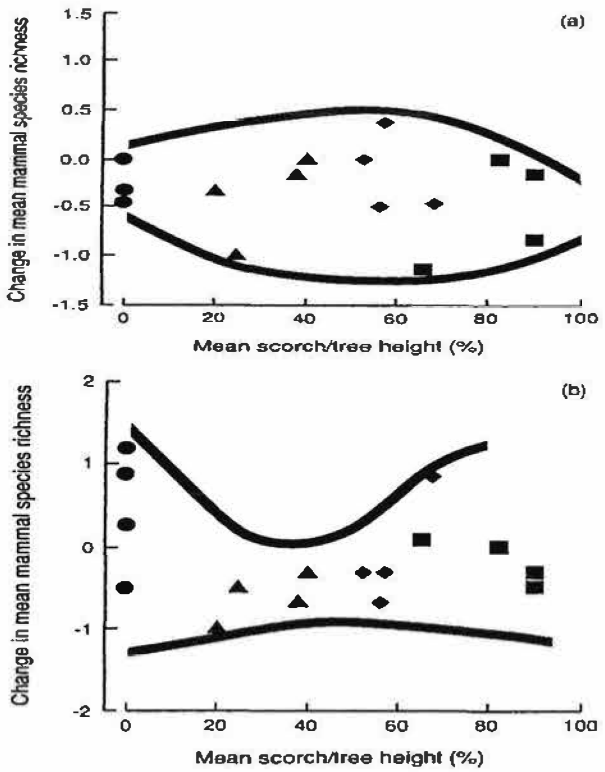

Figure 4. Change in (a) terrestrial and (b) arboreal mammal species richness in relation to mean scorch height/ tree height ratio for savanna vegetation for (a) ground and (b) canopy layer vegetation. Symbols as in Fig. 2.

Relationship between Vegetation Diversity and Mammal Abundance and Richness

Before the experimental fires in 1990, the relationship between total mammal abundance and vegetation patchiness and between mean species richness of mammals and vegetation patchiness was positive and was significant in five out of twelve cases, mainly with the tree layer (Table 1). However, after the commencement of the experimental fires the simple relationships disappeared. The fire regimes appear to have much less impact on the relationships with the slower variable of the canopy heterogeneity (>3 m, Table 1).

Change in Mammal Richness

The patterns of change in mammal species richness are similar to those obtained with plant richness and vegetation heterogeneity (Fig.4). Change in terrestrial mammal richness is relatively minor at low and high intensities and greatest around 60 per cent scorch/ height. Change in arboreal mammal richness again shows similarity with earlier canopy patterns, with greatest change at low and high intensities and least change at intermediate intensities.

DISCUSSION

The strong positive relationship between mammal richness (diversity) and vegetation patchiness was lost for terrestrial mammals immediately the fire regimes were imposed. The effects of fire on the ground vegetation were immediate and the impact on terrestrial mammals was similar. It is suggested that in the absence of fire the mammal numbers on the burnt sites would not quickly respond as they appear to have on the Natural sites. It must be remembered that for two years before the fires started in 1990, all sites were unburnt. Thus it would appear that longer-term spelling (more than a couple of years) for some places may be required to see such an impact.

These preliminary data provide some support for the idea that a complex regime consisting of patches burned by a range of different fire types ultimately drives habitat heterogeneity which is, in turn, important for the mammals. The canopy and ground layers certainly behave differently. They represent slow and fast variables. The ground layer responds most strongly to the intermediate fire intensities which are also the most spatially variable. This part fits the conventional wisdom. The patchy early dry season fires are

TABLE 1

Correlation coefficients between mammal abundance and richness and ground and tree layer heterogeneity before and after experimental fires on Kapalga. * $p<0.05$ ** $p<0.01$ *** $p<0.001$.

MAMMALS	HETEROGENEITY: GROUND LAYER ABUNDANCE	GROUND LAYER RICHNESS	TREE LAYER ABUNDANCE	TREE LAYER RICHNESS
PRE-FIRES (1990)				
All	0.430	0.304	0.802***	0.739***
Terrestrial	0.103	0.678**	0.416	0.568*
Arboreal	0.356	0.440	0.562	0.771***
POST-FIRES (1991)				
All	0.212	0.245	-	-
Terrestrial	0.232	0.221	-	-
Arboreal	0.224	0.068	-	-
POST-FIRES (1992)				
All	0.105	0.304	0.239	0.592*
Terrestrial	0.064	0.197	0.161	0.584*
Arboreal	0.127	0.142	0.250	0.094

pragmatically convenient for breaking up the country in order to prevent the spread of later, more intense fires. This enhances property protection and is intuitively appealing as a conservation measure. It increases habitat heterogeneity and limits the largely presumed destructive effects of late fires. Most importantly, some areas are also protected from burning. These patches not only contribute to habitat heterogeneity but allow recruitment into the canopy from the ground layer.

In northern Australia, the present biodiversity of the savanna landscape appears to be maintained by an historic anthropogenic fire regime. It is probable that an approximation of the landscape pattern produced by the traditional Aboriginal burning regime maximizes biodiversity through maintaining habitat diversity, savanna patchiness and species diversity and protecting endemic species. The research task is to determine and optimize that fire mosaic.

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APPENDIX 1

List of mammal species caught during study classified into arboreal and terrestrial, and marsupial and native rodent categories.

STRATUM HABIT	MARSUPIAL	RODENT
Arboreal	*Dasyurus hallucatus*	*Melomys burtoni*
	Antechinus bellus	Mesembriomys gouldii
	Phascogale tapoatafa	
	Trichosurus vulpecula	
Terrestrial	*Sminthopsis virginiae*	*Pseudomys delicatulus*
	Isoodon macrourus	*Pseudomys nanus*
		Rattus colletti
		Rattus tunneyi

CALMScience Supplement 4: 197-198 (1995)

Fire management in habitat islands: risks and impacts - a fauna perspective (abstract only)

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ABSTRACT

In the high production agricultural regions of South Australia native vegetation is extremely fragmented and frequently highly modified. Fire is regarded as both the major threat and the major habitat management tool in these isolated remnants. Two case histories are discussed which firstly illustrate the risks from wildfires, and secondly demonstrate the impacts of prescribed burning frequency on regeneration of the overstorey and its habitat value for hollow-dependent fauna. These case studies are used to discuss fire management issues in habitat islands. In areas subdivided by access tracks each block has the potential to be managed as a mosaic within a mosaic, but in the absence of internal access, use of fire for habitat management requires a sound understanding of fire behaviour, and a flexible ignition technique.

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Mosaics in Sydney healthland vegetation: the roles of fire, competition and soils

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ABSTRACT

Sydney's coastal sandstone plateaux support a heathland mosaic that includes thicket of tall dense shrubs and heath with a diverse complement of small shrubs, forbs and graminoids. Thicket dominants are infrequent or absent in heath, while smaller plants that typify heath are less frequent and less diverse in thicket. Examination of a sequence of aerial photographs spanning 50 years showed that the mosaic was dynamic over time and that changes in the distribution and abundance of the two structural forms are related to fires. Comparative studies of *Banksia oblongifolia*, a species of understorey shrub, have shown that its fruit production is reduced when in the presence of mature thicket dominants. This species, and numerous others with similar life-cycle attributes, is less abundant in thicket than in open-heath, suggesting that competition from thicket dominants may be an important factor influencing community composition. Manipulative experiments to test this hypothesis are in progress. Analyses of soil chemistry showed some variability between sites within the mosaic. There was evidence that some aspects of soil chemistry are dynamic through time in response to temporal changes in vegetation.

Fire management of mosaics must take cognisance of complex interactions that control spatial and temporal variation. If full diversity is to be conserved, a variable fire regime must be implemented that is responsive to the existing state of the system and its rate and direction of change.

INTRODUCTION

Ecosystems are traditionally depicted as a web made up of interactive components. Agricultural systems may be managed by manipulating any of these components, but in natural systems interactions, complexity and cost dictate that fire is the principal management tool. We want to manipulate natural systems through fire management to conserve biodiversity and to protect life and property. Socio-political pressures often lead managers to consider regular pre-defined fire regimes directed principally at fuel management, or regimes designed with the intention to protect one or a few species perceived to be important because of rarity or socio-economic value. This paper evaluates the effectiveness of such management strategies in the maintenance of full biodiversity by examining patterns and dynamics in heathland mosaics near Sydney.

Study Area and Methods

Heathlands on Sydney's coastal sandstone plateaux comprise mosaics of dense shrub thicket and open heath/sedgeland. Thicket (Banksia Thicket of Keith and Myerscough 1993) is dominated by an overstorey of tall serotinous obligate seeding shrubs (e.g. *Banksia ericifolia*) with an understorey of smaller shrubs (e.g. *Pimelea linifolia*), graminoids (e.g. *Lepidosperma neesii*) and forbs (e.g. *Goodenia dimorpha* var. *angustifolia*). Open heath (Restioid Heath of Keith and Myerscough 1993) is without overstorey, but has a more diverse array of small shrubs, graminoids and forbs than thicket. The study area is 350 ha of heathland west of Jibbon Hill (34° 09'S, 151° 09'E) in Royal National Park.

Fires and the two vegetation types were mapped over a period of 50 years using a sequence of aerial photographs. The maps were digitized and incorporated into a geographic information system (GIS) based on 25 m grid cells, allowing the distribution of thicket to be compared between times and related to the occurrence of fires. Field studies on plant populations and soils were carried out after a fire in 1988. Samples were located along transects in each of open heath and several structural variants of thicket characterized by differences in height and density of dominant plants (Keith and Bradstock 1994).

RESULTS

Fire

Management of the mosaics centres on whether they are static and related to physical environmental factors or spatially dynamic in response to temporal events such as fire. Figure 1 clearly shows that spatial patterns are dynamic. There was no evidence of thicket in the area prior to 1960 (maps not shown). The extent of thicket increased at the expense of heath from 1960 through to the mid 1970s, when a major reduction occurred. Thicket again increased from the late 1970s until 1988. After 1988 another major reduction in thicket occurred (map not shown).

Dynamics of the mosaic are related to fire (Fig. 2). The overstorey of thicket was destroyed by fire (at least temporarily). In the absence of fire, the extent of thicket increased. The gradual, rather than sudden post-fire expansion was not due to continuous seedling recruitment in the years after fire. Rather, it reflects spatially variable growth rates that cause thicket dominants to become visible on aerial photographs at different times since fire.

Banksia ericifolia is the major overstorey species in thicket. The population model developed for *B. ericifolia* by Bradstock and O'Connell (1988) is applicable to other overstorey species (*Hakea teretifolia* and *Allocasuarina distyla*) because these have similar life histories. The model identifies fire frequency and seedling establishment as the major factors controlling population density. If fires recur at less than a critical interval, then overstorey populations will decline. The length of the critical interval varies between six and 13 years depending on the level of seedling establishment, which is a function of post-fire rainfall and site quality (Bradstock and O'Connell 1988).

The model predictions were examined at landscape scale by analysing the history of fire, fate of thicket and rainfall records between 1972 and 1988. In the interval between 1972 and 1988, three fires burnt various parts of the study area, hence there were eight possible fire histories (Fig. 3). Four of these (BBB, BNB, NNB and NNN) were spatially restricted and therefore not analysed. Two of the remaining fire histories (BNN and NBN) had one fire between 1972 and 1988 and two (BBN and NBB) had two fires. The penultimate fire intervals for BBN and NBB were 2.5 years and 4 years, respectively (i.e. less than the critical interval of Bradstock and O'Connell 1988). BNN and NBN had penultimate fire intervals of at least 10 and 12 years, respectively (i.e. more than the critical interval unless seedling establishment was minimal). In the two years after fire, annual rainfall was highest after the 1974 fire, lowest after the 1980 fire and intermediate after the 1976 fire (Fig. 4).

At both 1972 and 1988, the minimum post-fire age of any grid cell was 8 years. It was assumed that grid cells in which no overstorey was visible on aerial photographs flown in 1972 and 1988 did not contain overstorey in an immature (hence undetectable) state. Grid cells that contained overstorey in 1972, but not in 1988 were recorded as overstorey extinctions.

The proportion of overstorey extinctions was greater in grid cells subject to short fire intervals (BBN and NBB) than in grid cells subject to fire intervals as long or longer than the critical interval (BNN and NBN) (Fig. 5, $P<0.001$). The effect of post-fire rainfall was examined by comparing BBN with NBB and BNN with NBN. In each case, where rainfall after the last fire was lowest (NBB and NBN, respectively) there was a higher proportion of extinctions (Fig. 5). The differences in frequencies of patch extinctions observed in relation to fire frequency and post-fire rainfall suggest that Bradstock and O'Connell's (1988) population model is applicable at landscape scales.

The results showed that a single short fire interval is not sufficient to drive overstorey to widespread extinction. In BBN, the short fire interval did not cause overstorey extinction in all grid cells (even though recruits after the 1974 fire could not have produced seed before being killed by the 1976 fire). This was possibly because: (i) a small residual seed bank may be retained in cones of dead plants, allowing an opportunity for recruitment after a second fire (Bradstock, unpublished); and (ii) seeds may be dispersed from nearby patches that did not experience a short fire interval.

Competition

Richness of understorey species, particularly shrubs, varies inversely with overstorey density (Fig. 6). This relationship has been observed in other heathlands and has been put forward as evidence that overstorey adversely affects understorey through competition (e.g. Specht and Specht 1989; Cowling and Gxaba 1990). The mechanism of such an interaction is that dense overstories deprive understorey plants of resources, reducing their survival and reproduction, and eventually cause elimination (Keith and Bradstock 1994). The process is interrupted by fire which removes overstorey, either temporarily until it regains dominance, or, in the case of frequent fire, overstorey cover may be reduced or eliminated for one or more fire intervals.

Understorey species vary in their response to overstorey competition. This was examined by defining functional groups of species based on their life-history attributes. One group consists of serotinous resprouters (e.g. *Banksia oblongifolia*), while a second group includes obligate seeders with longer lived soil seed banks (e.g. *Acacia suaveolens*). Observations on population densities suggest that all eight species surveyed in the resprouting group were adversely affected by various overstorey types (Table 1). In contrast, only one of eleven species in the obligate seeder group had the same response, while the remainder were either unaffected by overstorey or responded in a way that could not readily be ascribed to competitive effects.

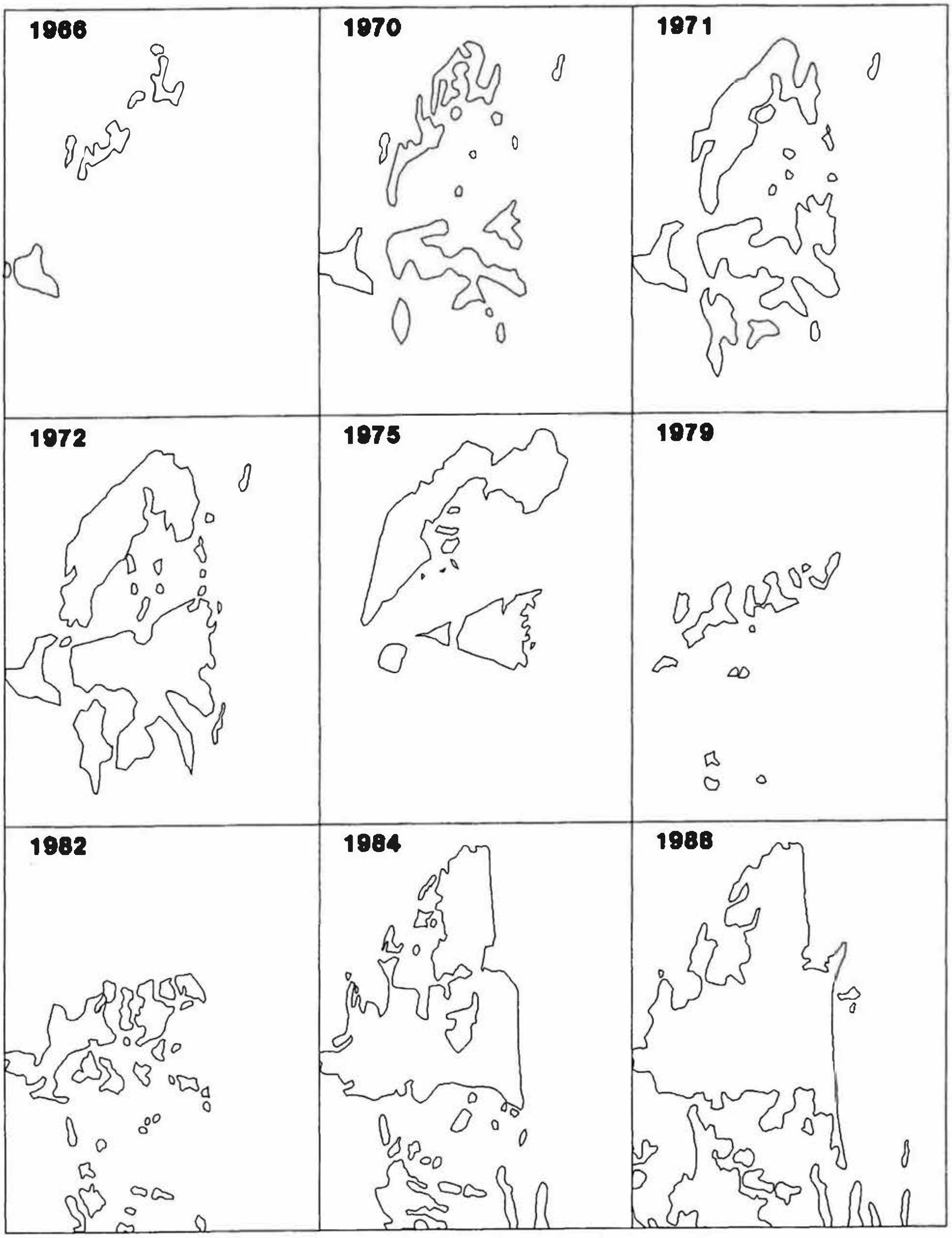


Figure 1. Series of maps showing changes in the distribution of thicket 1966-1988. The remaining area in each map is open heath.

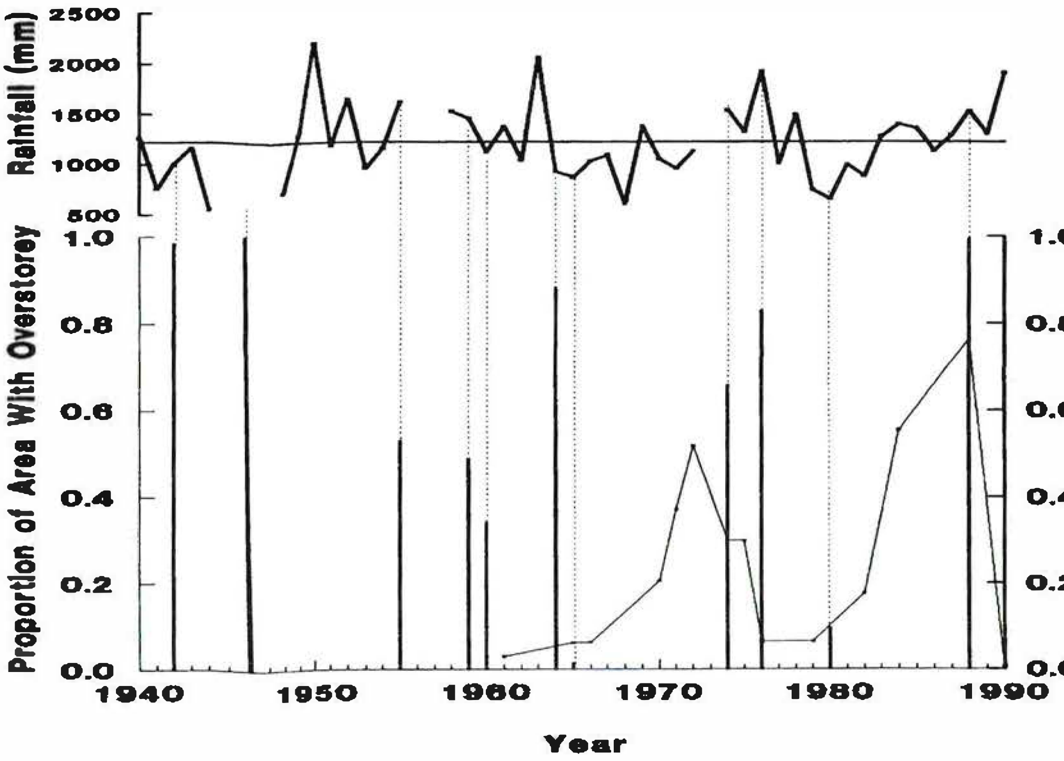
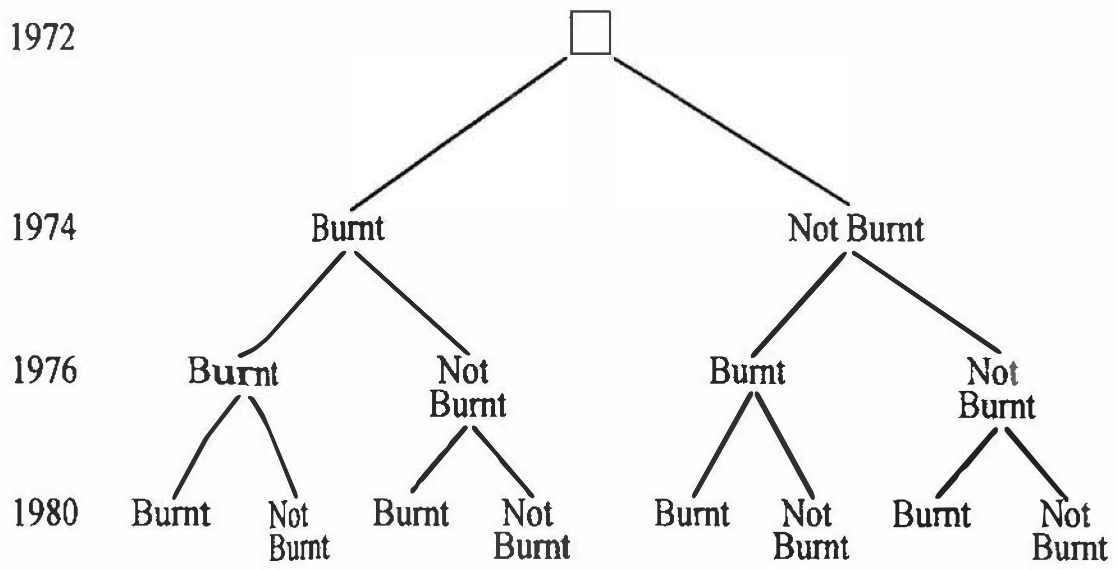


Figure 2. The extent of thicket (thin line) and fires (unbroken vertical lines) for the period 1940-1990. Increases in overstorey coincide with intervals of more than 10 years between major fires. Declines in overstorey coincide with major fires in 1974-1976 and 1988. Top graph shows variation in annual rainfall, horizontal line represents mean.



	(BBB)	(BBN)	(BNB)	(BNN)	(NBB)	(NBN)	(NNB)	(NNN)
Cells	7	265	3	92	39	154	0	2

Figure 3. Tree diagram showing eight possible fire histories between 1972 and 1988 and their extent (number of grid cells burnt).

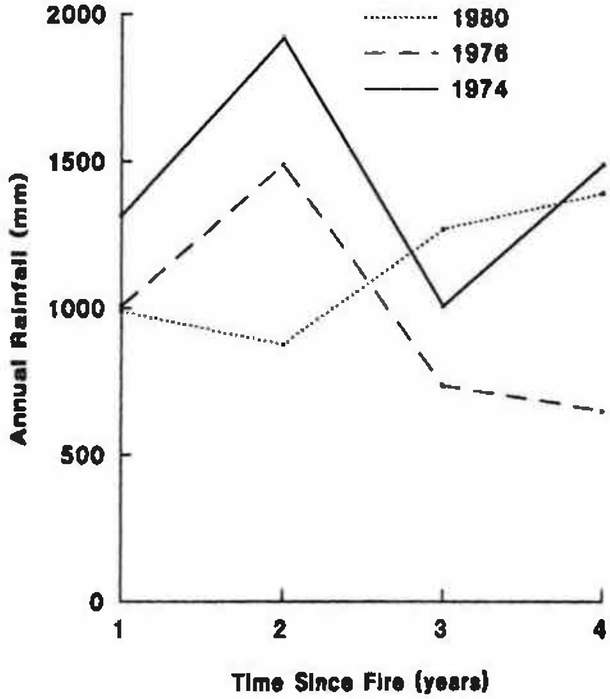


Figure 4. Annual rainfall after fires in 1974, 1976 and 1980.

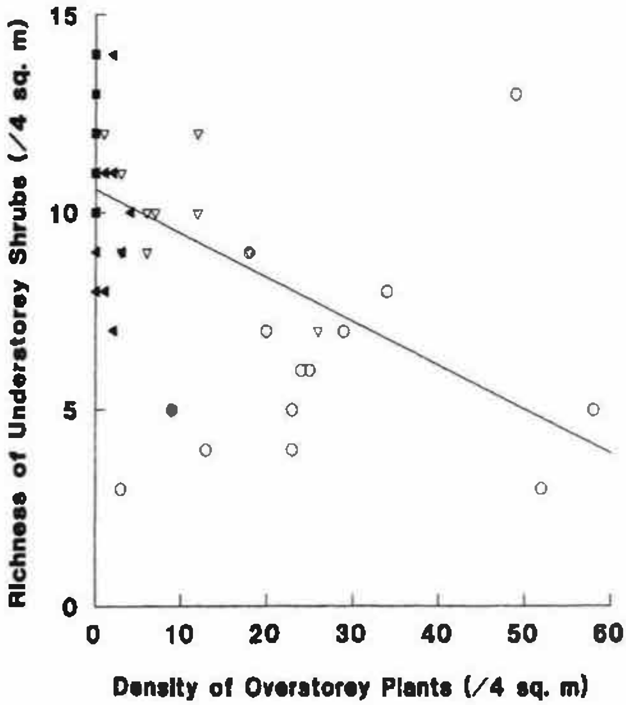


Figure 6. Relationship between overstorey density and richness of understorey shrub species, $R^2=0.27$, $P<0.001$ (after Keith and Bradstock 1994).

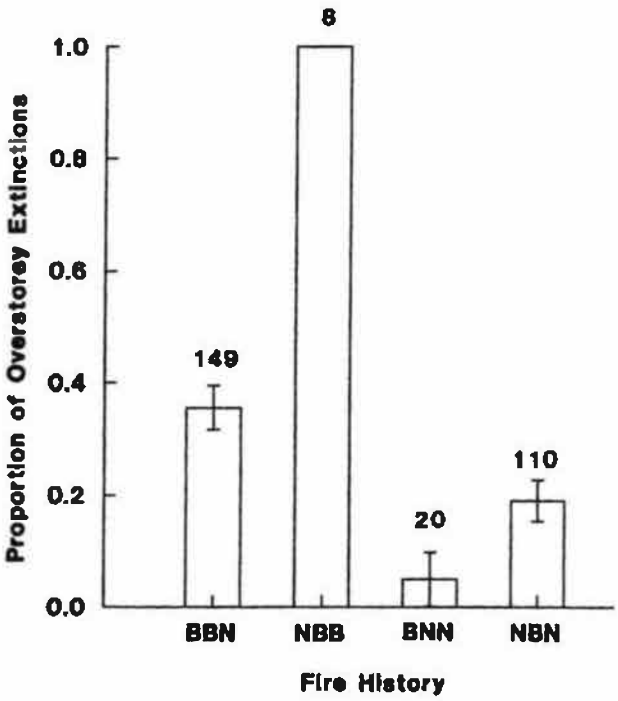


Figure 5. Proportion of overstorey extinctions under four fire histories. Numbers are total grid cells with overstorey in 1972 for each fire history. Error bars show standard errors.

TABLE 1

Differences in density of post-fire populations in two groups of understorey shrubs: serotinous resprouters and obligate seeders with soil seed banks. Overstorey types: A- absent; O- open; S- short; T- tall.

SEROTINOUS RESPROUTERS	
Allocasuarina nana	A>S O T
Banksia oblongifolia	A>O>S>T
Callistemon linearis	A O S T
Hakea dactyloides	A>S O T
Isopogon anemonifolius	A>O S T
Lambertia formosa	A>S O T
Leptospermum continentale	A>O>S T
Melaleuca nodosa	A>T O S
OBLIGATE SEEDERS WITH SOIL SEED BANK	
Acacia suaveolens	ns
Cryptandra ericoides	T>S O A
Dillwynia floribunda	O>T>S A
Epacris microphylla	O A>T>S
Eriostemon buxifolius	ns
Gompholobium glabratum	T A>O S
Leucopogon microphyllus	O>T S A
Mirbelia rubiifolia	A>O S T
Persoonia lanceolata	ns
Pimelea linifolia	O>A S>T
Xanthosia tridentata	ns

Differences between the two functional groups relate to their ability to exploit a post-fire window of reduced overstorey competition (Keith and Bradstock 1994). In the obligate seeder group, post-fire recruits mature rapidly and establish a long-lived seed bank before overstorey species regain their dominance and exert competitive effects. Post-fire recruits of resprouters are unable to reproduce within the window due to their slower growth and maturation. Beneath overstorey, they may suffer increased mortality and/or reduced fruit production. If recruitment fails to compensate for adult deaths, populations will decline.

Observations on *B. oblongifolia*, a serotinous resprouting understorey shrub, support this model. Its fruit production was reduced beneath or in the vicinity of various overstorey types (Fig. 7, $P<0.0001$), an effect that was translated into reduced recruitment in the next fire interval ($P<0.05$).

Soils

Soil properties vary spatially and temporally between vegetation types within the mosaic. Levels of topsoil nutrients are generally higher under various types of overstorey than in open heath (Fig. 8). Topsoil nutrients also increase in the early post-fire years, although organic matter decreases. There is evidence from a manipulative experiment that soil properties vary through time in relation to changes in overstorey. After fire, exchangeable cations accumulate in topsoil more rapidly where overstorey plants were retained than where regenerating overstorey was removed (Fig. 9, $P<0.01$). However, changes in organic matter content and some other soil properties occurred at a similar rate in treatment and control plots (Fig. 9, $P>0.05$).

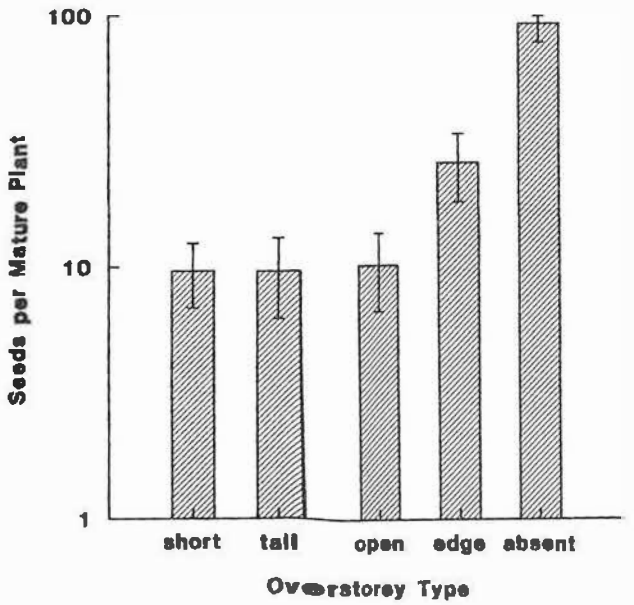


Figure 7. Fruit production of Banksia oblongifolia *in the presence of various overstorey types: short, tall and open are structural variants of thicket; absent is open-heath with no overstorey dominants; edge is boundary area between thicket and open heath (after Keith and Bradstock 1994).*

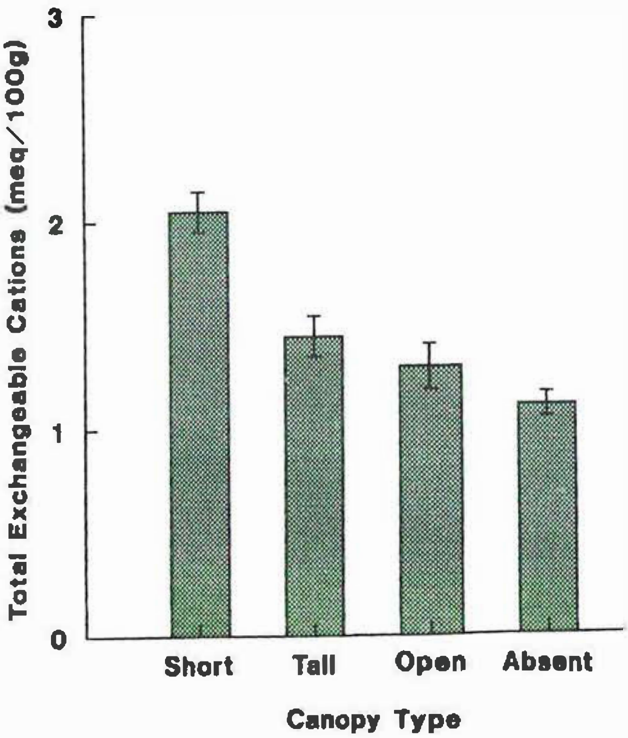


Figure 8. Total exchangeable cations in top soil (0-7 cm depth) beneath various overstorey types.

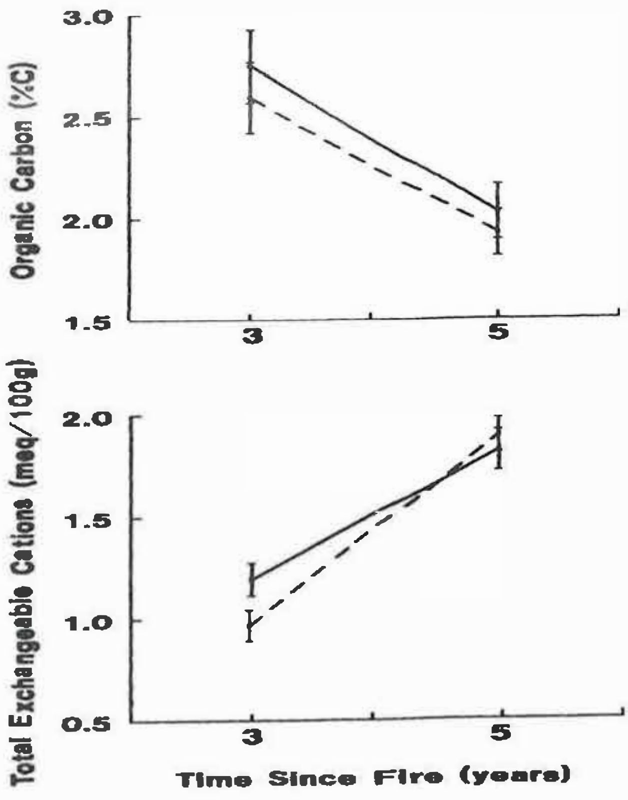


Figure 9. Rates of change with time since fire in organic carbon and total exchangeable cations of topsoil beneath a developing overstorey (broken line) and at sites from which seedlings of overstorey species were experimentally removed.

DISCUSSION AND CONCLUSIONS

Heathland mosaics are interactive and dynamic systems. The properties that characterize mosaics: spatial heterogeneity; temporal dynamics; and interactive processes, are inherent in all natural systems and are the ultimate means of sustaining their biodiversity. Mosaics should therefore be considered the norm of natural systems, rather than the exception. Management of these systems for conservation of their biota must focus on fire as the major component amenable to practical manipulation. Direct management of other components is either prohibitively expensive or untenable because of logistic limitations or a lack of knowledge of likely responses.

Fire management of heathland mosaics for conservation should aim to promote coexistence by avoiding at least two mechanisms of extinction: elimination of plants by frequent fire; and competitive elimination of understorey plants by overstorey over a series of long fire intervals. While mechanisms of animal extinction need to be identified and avoided in management, maintenance of structural and floristic diversity of vegetation may be the first step toward animal conservation.

A range of fire management options are available for conservation of heathland mosaics, for example (i) repeated application of frequent fires; (ii) repeated application of fires at intermediate intervals; (iii) a compartmental strategy where different fire regimes are applied to different patches; or (iv) a flexible strategy that seeks to implement reversible changes. Strategy (i) will cause decline and eventual extinction of overstorey species (and some understorey species), decline of dependent fauna (e.g. nectar feeding mammals and birds) and decline in certain soil resources. Strategy (ii) may avoid these effects and has some theoretical basis in the intermediate disturbance hypothesis of maximum diversity (Connell 1978), but will nonetheless cause decline and eventual loss of some understorey through overstorey competition. Strategies that are devised for management of single species or groups of species are of a kind exemplified by (i) and (ii): they identify a pre-defined schedule of fires that are perceived to be favourable or unfavourable to target species. These strategies will ultimately fail to meet broader conservation goals because they are 'blind' to dynamic interactions between components of the system that maintain its overall diversity.

Strategy (iii) is an attempt to resolve apparent conflicts in requirements of different species, by assigning them to specially managed patches. However, this static view of a mosaic is also unlikely to succeed because it ignores the temporal dimension of interactions and the vagaries of unplanned fires. The most appropriate strategy is a flexible one (iv) that is responsive to the existing state of the system and its direction and rate of change. Such a strategy is likely to result in a fire regime that is variable in space and time. The goal is to ensure that changes are reversible, rather than seeking to prevent change. Long or short fire intervals may need to be implemented over part of a mosaic from time to time to avoid spatially extensive extinctions. For example, a single short fire interval may release understorey species from suppression and is apparently insufficient to drive obligate-seeding overstorey species to extinction. Although the latter may decline, they may recover if subsequent fire intervals are long and the area affected by the short interval was not overly extensive.

Both strategies (iii) and (iv) must explicitly consider the scale fire mosaics. Mosaics are often put forward as a panacea for fauna conservation, but may fail if implemented at too fine a scale. Minimum viable habitat patch sizes and the concentration of predators on small patches may set a lower limit on the scale at which mosaics benefit biodiversity.

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A simple test fire exercise for fire behaviour training

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Abstract

Three advanced fire behaviour courses were held in New Zealand during 1992-93. We experimented with the idea of using small outdoor test fires (1.2 x 2.4 m) to help reinforce some of the principles and concepts being described in the classroom (e.g. fire development from point versus line ignition, influence of slope steepness, documentation of fuels, weather and topography in relation to quantified fire behaviour, meaning of certain fire behaviour characteristics). The fuel beds used consisted of forest floor material collected more or less *in situ* from beneath nearby radiata pine plantations and stored under cover for at least a month. Three test fires were undertaken simultaneously: point-source ignition at 0° slope, line-source ignition at 0° slope, and line source ignition at 10° slope. Course participants found the test fire exercises an invaluable element of the courses. For fire behaviour training purposes, the present methodology is an acceptable alternative to conducting field-scale test fires which are often difficult, if not impossible, to carry out in most course situations.

A time-dependent model of fire impact on fruits

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Abstract

Many plants rely on the seed in woody fruits for their post-fire regeneration. Therefore, seed survival during fire is critical. A model for the survival of seeds in woody fruits is constructed using heat-flow equations with time-dependent temperature inputs. The model is used to predict the survival of seed in fruits exposed to both laboratory heating and field fires as reported in the literature. The inclusion of thermal arrest in the inputs to the model gives the upper bounds for estimated times of seed survival. The model gives reasonable predictions of seed fate. It is shown that seed location in the fruit is not a critical factor provided the seed is within the central core of the fruit. The applicability of the model is also demonstrated using time-temperature curves at two heights from experimental fires burning in different fuel types.

Fuel modelling and fire behaviour in buttongrass moorlands

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Abstract

Buttongrass moorlands are a low open sedge community that covers vast tracts of western and south-western Tasmania. These moorlands have the interesting fire management characteristics of being highly flammable, overlaying peat soils, occurring in extensive unbroken plains and abutting fire sensitive vegetation. Management of both wildfires and fuel reduction burns has been fraught with problems and arguments. The Fuel Characteristics and Fire Behaviour in Tasmanian Buttongrass Moorlands research project has attempted to quantify the basic fire parameters to enable an objective approach to fire management in this fuel type.

Fuel characteristics in Tasmanian buttongrass moorlands have been sampled for a wide range of sites in south-western and western Tasmania. The fuel characteristics sampled were: various Rothermel fuel characteristics, fuel moisture, fuel high heat contents, fuel loads and percentages of dead to live fuel. Most of the variation observed of fuel loads can be accounted for in the variables geology, vegetation age and vegetation cover. Due to problems with measuring vegetation cover, age and geology are used to predict fuel loads. The percentage of dead fuel at a given age did not vary between different geologies, and is modelled using age alone. Two fuel load and one dead-fuel percentage prediction models have been produced. Only a preliminary fuel moisture model is available at present.

Forty-nine fires, including both research burns and wildfires, were measured for fire behaviour model development and a further seven fires used for model verification.

Empirical models have been produced to predict buttongrass moorland headfire rate of spread and flame heights, using the variables moorland age, dead fuel moisture and surface wind speed. Alternatively, fire behaviour predictions can be made from the moorland age, relative humidity, temperature and surface wind speed.

The rate of spread model takes the form of:

ROS = constant x wind function x moisture damping x fuel function.

The flame height model takes the form of:
FH = constant x (heat content x fuel consumption x rate of fire spread) ^{power}.

The models should provide good predictions for relative humidities between 30 and 100 per cent, temperatures between 8° and 25°C, and surface wind speeds below 15 kmph (i.e. low to moderate intensity fires). When wind speeds are between 15 and 35 kmph the models should provide adequate predictions.

Nitrogen budgets in regrowth karri following thinning and fuel reduction burning

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[2] Science and Information Division, Department of Conservation and Land Management, Manjimup 6258, Western Australia.

Abstract

The Department of Conservation and Land Management is undertaking an experimental program to investigate the feasibility and impacts of burning slash fuels created by thinning operations in young regrowth stands of karri (*Eucalyptus diversicolor*). Potential constraints on prescribed burning of thinning slash include damage to retained crop trees and losses of nutrients, particularly nitrogen, from the ecosystem. The amounts of nitrogen volatilized during fuel reduction burning of thinned karri forest are significant and for the most intense of eight experimental fires corresponded approximately to the amounts of nitrogen in growing vegetation (about 180 kg ha^{-1}). Compared with the total nitrogen in the ecosystem (vegetation + litter + soil) these amounts are small so that a single fire event is not likely to have a large impact on total nitrogen stores. However, inputs and outputs of nitrogen as a result of regular burning will affect the balance of nitrogen in the long term. Minimizing volatile losses of nitrogen will help to maintain this balance. This is best achieved by burning under conditions when the lower part of the fuel profile is moist (litter profile moisture content > 80 per cent), resulting in reduced combustion of the litter layer - the main compartment of nitrogen storage in fuel. Such conditions are also likely to minimize damage to retained crop trees.

A climatology of *Very High* and *Extreme* fire weather days in southern Western Australia

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Abstract

Days of EXTREME and VERY HIGH fire danger for the 22 seasons 1970-71 to 1991-92 (November to April inclusive) were identified using three-hourly data from the Bureau of Meteorology stations in southern parts of the State. Calculations of the fire danger index (FDI) were made using the McArthur Mark IV grassland fire danger meter assuming average fuel amounts and 100 per cent fuel curing.

The average number of EXTREME (FDI > 49) days per season graded from about one in the south-west corner of the State to more than 15 in the Eucla and around Geraldton, with appreciable year-to-year variability. In south-east parts of the State most EXTREME days occurred during the first half of the season whereas in the Geraldton area the majority of these occurred during January and February. Active fire seasons in the Geraldton region tended to correspond to less active seasons at Esperance and vice-versa. An analysis of VERY HIGH and EXTREME fire danger days (FDI > 26) during the period revealed a similar distribution but values ranged from an average number of 10 days per season in the south-west corner to about 50 days at both Forrest and Geraldton. The monthly distribution of these events is also similar.

A study of weather patterns that led to the EXTREME fire weather conditions revealed that the pre-frontal trough was the major influence in south coastal areas whereas the combination of a strong high to the south of the State and a trough over the Gascoyne was dominant for west coastal areas. Strong afternoon sea breezes due to a deepening trough inland and a high to the west of the State also generated EXTREME conditions along the Geraldton coastline.

Fire in the human ecosystem - a question map

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Abstract

The human ecosystem consists of human beings and their interactions with their environment. The human environment is a seamless garment, but for conceptual

purposes can be split into two parts. 'Society' is the part of our environment, created by us, which would vanish with us were we to disappear in a puff of smoke. 'Nature' is the part of our environment which would remain. Human artefacts lie somewhere in between, since they would outlast us for a while, but eventually succumb to entropy due to a lack of maintenance. Our survival and wellbeing depend upon our activities within both parts of our environment, so that CALM policy must take both into consideration, seeking harmony between human systems and natural systems. As a human activity, prescribed burning is linked to many phenomena, both natural and social. Question-maps are a way of starting to think in a structural way about our activities, our wellbeing, society and nature, and a question-map about fire in our forest systems leads us into conceptual model building, or Systems Ecology, using ideas from General Systems Theory, Graph Theory, Matrix Theory, and Human Ecology. The resulting models are useful for policy formulation, management training, public education and research project management.

Fire history mapping of remnant urban bushland near Perth, WA

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Abstract

The fire history since 1948 at Star Swamp Bushland Reserve is interpreted from aerial photographs. Fire scars were traced separately from the 18 sets of aerial photographs available for the period. Fire frequency and the number of months since fire, were calculated for 185 survey points used for vegetation mapping as well as 32 permanent plots used for monitoring vegetation dynamics. The effect of fire frequency on the canopy of the dominant tree species *Eucalyptus gomphocephala,* composition and distribution of plant communities, and weed invasion was analysed using the geographic information system ARC/INFO, correlative and multivariate techniques. It is shown that mapping fire history can be a valuable aid for the management of urban reserves requiring special fire protection.

Planning for fire management in a near-urban national park

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Abstract

John Forrest National Park is located on the edge of the Darling Scarp 25 km from Perth. It is almost totally surrounded by residential areas. The major objective in planning for fire management is to protect life and property without compromising environmental and ecological values.

The fire plan proposed in the draft management plan for John Forrest National Park (Plan A on poster) was amended after considering submissions that indicated the draft proposal did not provide sufficient protection for the community from wildfire. A series of alternative suggestions was submitted during the public review process (Plans B, C and D on poster). A revised fire plan (Plan E) was developed, taking into consideration these options, in consultation with people who contributed submissions to the draft plan for the Park.

The fire management plan aims to provide the greatest diversity possible within the constraints of protection of life and property. Fuel Reduction areas will be reviewed annually in the light of additional scientific knowledge, and the effect of unplanned fires to determine whether or not they should be burned for ecological or protection purposes.

Patterns of resprouting of eucalypts after fire

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Abstract

Patterns of resprouting of *Eucalyptus rossii* and *E. macrorhyncha* were monitored on two sheltered and two exposed aspects following a wildfire in February 1991 in the Black Mountain Nature Reserve, ACT. Overall 673 individuals were monitored, 87 per cent of which resprouted. Trees started sprouting 54 days after the fire although sprouting of *E. macrorhyncha* was delayed for up to two months on exposed aspects. Sprouting continued throughout winter, however, rates

appeared to 'slow down' on sheltered compared with exposed aspects over the cooler months. Most trees had sprouted within a year of the fire; three individuals were recorded sprouting after October 1992 (20 months post fire). Larger individuals sprouted more rapidly - approximately 90 per cent of individuals ≥20.1 cm d.b.h had sprouted by the end of August 1991 compared with less than 50 per cent of individuals <20.1 cm d.b.h. The percentage of individuals sprouting only from the base of the tree increased over time. Most of the later sprouters were smaller trees with complete cambial death on the stem. Patterns of resprouting were related to, at least, soil moisture availability, air temperature and tree species, size and vigour.

Pre-European fire history of North American tallgrass prairie

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Abstract

Fire was an important component of the historical maintenance of the North American Tallgrass Prairie. Three aspects of the natural (pre-European) fire conditions have recently been studied. First, fires were estimated to have occurred an average of every 4.8 + 0.56 years based on fire-scarred trees growing along the margin of extant native prairies. Second, while not optimal for present objectives such as domestic livestock, fires occurring during the growing season may have been important in maintaining the natural diversity of the ecosystem. The herbs, false sunflower (*Heliopsis helianthoides*) and white aster (*Aster ericoides*), for example, increase with summer and fall burning; big bluestem (*Andropogon gerardii*), the dominant grass of the tallgrass prairie, responds best to spring burning. Fires routinely applied during the same season ultimately may reduce ecosystem diversity. Thirdly, simulated grazing designed to approximate the effect of large grazers such as bison (*Bison bison*) on fuel distribution, resulted in a significantly greater fire temperature heterogeneity ($P<0.001$) than occurred without grazers. Fire in the pre-European tallgrass prairie thus appears to have been a complex factor involving frequent and seasonally-variable occurrences and heterogeneous fire-temperatures across a grazed landscape.

Fire frequency and floristic variation in dry sclerophyll communities

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Abstract

Fire frequency, a function of the number of fires experienced by a community within a given time period, may be resolved into the components of time since the most recent fire and the lengths of intervals between fires. The dynamics of dry sclerophyll woodlands in the Sydney region were examined in relation to fire frequency in the recent (< 30 years) fire history. Direct gradient analysis of floristic data indicates that:

(i) Fire frequency accounts for around 60 per cent of the floristic variation among the samples.

(ii) The effect of time since fire and the length of intervals between fires on floristic composition was equal in magnitude but unrelated in the nature of the variation associated with them.

Increasing time since fire is associated with a decline in the evenness of fire-tolerant species while inter-fire intervals of decreasing length are associated with the decrease in evenness of fire-sensitive species. Increasing variability of the length of the inter-fire intervals is associated with an increase in the richness of fire-tolerant and fire-sensitive species, implying that it may be variation of inter-fire intervals that is responsible for maintaining the presence of a wide range of plant species in a particular community.

Modelling the impact of fire on the population dynamics of the Splendid Fairy Wren

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Abstract

During a 20-year study of a population of Splendid Fairy-wrens near Perth, nine wildfires have impinged on the area. Although the fires did not directly affect the survival of wrens, they had a major effect on reproductive success in the following years. On average, 19 per cent of female-years experienced fire in the 12 months prior to nesting and 33 per cent of

female-years in the two years prior to nesting. The fires had a dual impact. Most importantly, the rate of nest predation almost exactly doubled in the years following fire and secondly the onset of breeding was delayed by up to a month, presumably because the wrens had trouble finding suitable vegetation in which to hide nests.

In addition to fire frequency and nest predation, population growth is influenced by brood parasitism, seasonal fluctuations and patch-size, presently a complex picture of demographic-environmental interactions. These data have been incorporated into a computer simulation model which can be used to make predictions of likely outcomes from a variety of landscape and management scenarios.

Influence of herbivores on the vegetation and fire fuels of the Perup Forest region of Western Australia

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Abstract

Studies in the southern jarrah (*Eucalyptus marginata*) forest situated in the Perup Nature Reserve indicated significantly higher cover values for plant species inside wire-mesh exclosures after 10 years compared with outside the exclosures. Particular species that were favoured by herbivore exclusion included *Bossiaea ornata, Billardiera variifolia, Opercularia hispidula, Logania serpyllifolia* and *Tetrarrhena laevis*, among others. Plant species showing the greatest decrease in cover outside wire enclosures were found in the faecal pellets of the herbivores of the region. Faecal analyses documented a preference for 42 forest species by the Western Grey Kangaroo (*Macropus fuliginosus*), the Western Brush or Black-gloved Wallaby (*Macropus irma*) and the Tammar Wallaby (*Macropus eugenii*). The Common Brushtail Possum (*Trichosurus vulpecula*) consumed not only leaves of the dominant trees, but sampled species from the understorey, including *Leptomeria cunninghamii* and *Hakea lissocarpha*. Faecal samples of the Western Ringtail Possum (*Pseudocheirus peregrinus occidentalis*) included only forest canopy species. Overlaps in the diets of herbivores indicated the possibility of competition for plant resources, but the polyphageous nature of all Perup Forest herbivores and an ability to shift resource preference would indicate the food resources are probably not limiting in this region of the forest despite some habitat fragmentation. The polyphageous nature of the native herbivores also indicates that rare plants are probably not endangered due to feeding effects by the animals. Herbivory has strong implications for fire management in the animals ability to reduce fuel loads and preferential feeding choices on fire-regrowth could affect particular species populations.

Influence of fire on the seed germination ecology of species of the jarrah forest

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Abstract

Plant species of the jarrah forest can be categorized by their life history syndromes related to the survival of fires and their mode of seed dispersal. Obligate seeding species require re-establishment following fire from seed because the parent plant is killed by the fire. Resprouting species differ in that the parent survives and reproductive output by seed is usually limited. Also, species may differ in the timing of seed dispersal; i.e. seed dispersed annually to the soil or retained in serotinous fruits for the plants for a period of years. Obligate seeding, soil seed store species, especially jarrah forest legume species, often have seed dormancy mechanisms which prevent them from germinating until after a fire. The heat shock provided by the fire can serve to break an impervious seed coat or possibly denature some seed coat inhibitor. To differentiate potential differences in these two influences of fire, differential germination results following scarification and boiling revealed that the jarrah forest could have both types of species. Examples of species which germinate following the mechanical breaking of the seed coat include *Acacia nervosa, Bossiaea eriocarpa, Daviesia physodes*, and *Gompholobium knightianum*. Species predicted, but not proven to have seed coat inhibitors are *Acacia drummondii, A. pulchella, Gastrolobium spinosum* and *Oxylobium cuneatum*. In addition to a requirement for a heat shock pre-treatment many jarrah forest species also must have the proper temperature and light cues to break dormancy. For example, *Acacia pulchella* var. *glaberrima* germinates in highest percentages corresponding to winter incubation temperatures in the dark, while *Banksia grandis* and *Hakea amplexicaulis* seeds germinated best at cool temperatures, but when light intensity is high. The implications for maintenance of the species under forest management are described.

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Survival of trapdoor spiders during and after fire

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Abstract

The response to fire by two myglamorph 'trapdoor' spiders with very different life histories was studied. The study site was in sandplain heath/shrubland in Durokoppin Nature Reserve in the central Wheatbelt of Western Australia. The fire, in March 1989, was part of several experimental fires undertaken by CSIRO and CALM to study the effects of fire on the biota as part of a program to assess the role of fire in management of small reserves.

The two trapdoor species studied were *Anidiops villosus* (Rainbow) and an unnamed *Cethegus* species.

Cethegus spiders are web weavers which catch prey in a flocculent, curtain-like web over a shallow retreat burrow (Main 1960, 1964; Raven 1984). Webs are sited against the base of small shrubs, logs or fallen branches. The species studied here matures rapidly (in about a year) but females continue to live for at least several years. Spiders may move nest sites a short distance if webs are damaged by heavy rain. Juveniles disperse aerially (Main 1991, and personal observation).

In contrast, *Anidiops villosus* digs a deep (70 cm), permanent burrow closed by a trapdoor and with a radiating fan of twiglines attached to the burrow rim (Main 1978). These twiglines are used for foraging. Nests are sited in litter under the the shade of shrubs and low trees. Spiders have a long developmental period (at least eight years for males and longer for females). Females reproduce iteroparously and may live for upwards of 25 years (Main 1987, and unpublished data). Spiders are dependent on both shade and permanent litter. Juvenile dispersion is ambulatory and restricted to a short distance, often to the litter mat of the maternal shade tree.

Webs and burrows were marked individually with numbered steel tags on wire pegs along two transects (625 m and 100 m long) in the site to be burnt and along a parallel transect (625 m) in an adjacent control (non-burn) site. Nests along all transects were marked progressively when found on census dates between 5 September 1987 and 17 December 1988. Nests viable at the last day of census marking were censused again on 18 March 1989 one week after the fire.

Response of *Cethegus*

All active nests at the last pre-burn census (17 and 1) along the burnt transects (625 m and 100 m respectively) were destroyed and the spiders presumed killed by the fire. Of the the 11 active nests at last pre-fire census in the unburnt control 10 were still active on the post-burn census date. By June and July 1989 and 2 February 1990 (eleven months post-fire), none of the burn site nests had recovered nor were there any webs rebuilt nearby. In the control non-burnt site, of the 11 nests active on the pre-burn census date, 10 were still active at the post-fire census date. During the autumn following the fire (1990), aerially dispersed spiderlings from adjacent unburnt bush recolonized the regenerating bush on the burnt site.

Response of *Anidiops villosus*

Most adult nests (11 of 14 on the long transect, 5 (all nests) on the short transect) survived the fire; 17 of 24 along the unburnt transect persisted (of these one had been preyed upon). On the burnt site four (of 11) and one (of 5) became defunct within three months following the fire. Several of these nests were vigorously attacked by birds, i.e. by pecking off rebuilt doors and twiglines thereby disrupting the spiders' foraging capabilities. There was no recruitment of juveniles in the burnt site during the autumn-winter following the burn.

Conclusions

From survival and recruitment data of *Cethegus* and *Anidiops* following an induced burn it seems that *Cethegus* although destroyed outright by fire is able to recolonize a regenerating post-burn site provided there is adjacent unburnt habitat with a reservoir population.

Conversely, although adult spiders of *Anidiops* survive fire, spiders are disadvantaged in a post-burn habitat due to inadequate shade, litter and possibly reduced prey and exposure to predation - all factors which lead to a progressive mortality following fire. Main (1978) suggested that the behaviour and deep burrows of the spiders (adaptations to aridity and drought) 'fits them to survive through a bushfire' and also that fire would probably not be deleterious to a population. However, although the observations reported here show that adult spiders can survive a fire, the post-fire mortality combined with lack of juvenile recruitment means that in a population sense *Anidiops* is indeed vulnerable to fire.

These contrasting responses by two mygalomorph spiders demonstate that a knowledge of invertebrate species' life history particularities is desirable in order to adequately manage small reserves for maintenance of their species diversity.

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Manuscripts must be on quality A4 white paper printed on one side of the paper only. The characters must be clear and black and all margins must be at least 3 cm. The pages of the manuscript must be numbered consecutively, throughout the entire paper, beginning at the title page, and including those pages containing references, appendices, tables, illustrations, captions, all of which are placed after the text.

Spelling should conform with the preferred (i.e. first-cited) spelling of the Oxford English Dictionary, with the single exception of the word 'program'.

SI units should be used for exact physical quantities and measurements. Authors who are unfamiliar with the SI units should consult AS 1000-1979 *The International System of units (SI) and its Application* issued by the Standards Association of Australia (1979).

Numbers referring to unit of measurement (e.g. 10 cm) are never spelt out unless forming the beginning of a sentence. All numbers under 10 not referring to units of measurement are spelt out.

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