

The use of fire for conservation management in New Zealand

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Abstract

In order to develop policy on the use of fire for conservation management in New Zealand, The Department of Conservation commissioned Landcare Research to provide a summary of the scope of previous scientific reviews of fire ecology in New Zealand, to review the overseas and New Zealand literature on the use of fire for management of grasslands, shrublands and wetlands for conservation purposes, to review techniques for monitoring fire impacts and recovery, and to make recommendations for use of fire in ecological management in New Zealand.

A literature review is presented along with recommendations on fire management and the use of prescribed fire in New Zealand protected natural areas.

Prescribed burning is used overseas as a management tool in fire-adapted mediterranean-climate ecosystems and some other grasslands and wetlands. Few New Zealand biota possess the adaptations found in plants and animals of ecosystems which have evolved in response to periodic fires. Although the use of prescribed fire could be considered to meet some management objectives in New Zealand protected natural areas, knowledge of its effects on many ecosystem components is scarce or lacking. Caution is advised in the planning and application of fire as a management tool.

1. Introduction

The Department of Conservation is working towards the development of a fire management policy that can be applied to the conservation estate in general, and to specific reserves. This will cover issues related to fire prevention and control, the use of fire as a conservation management tool, and restoration after fire. It would be applied to the management of both accidental fires that affect the conservation estate, and deliberate fires used for conservation management (referred to as prescribed fire).

Science and Research Division, Department of Conservation, requested Manaaki Whenua - Landcare Research to provide a review of the use of fire for conservation management, to review techniques for monitoring fire impacts and recovery, and to make recommendations for the use of fire in ecological management in New Zealand. The review was carried out in August and September 1995 jointly by staff from Landcare Research and Department of Conservation. This report presents results of the investigation and recommendations for the Department of Conservation to consider in the development of a fire management policy.

2. Background

"Since only the kiss of flame is needed to rouse dormant seeds from decades-long sleep, is it not strange that botanists do not turn arsonists on occasion that some floral phoenix might arise from the ashes?" (John Thomas Howell).

This quotation from a *Sierra Club Bulletin* of 1946 aptly demonstrates the difference in fire adaptation between the floras of North America and New Zealand. There is a considerable literature devoted to descriptions of adaptations to fire and the effects of fire on vegetation in North America, South Africa, Mediterranean Europe and Australia, where frequent natural fires have shaped the biota for millenia. In contrast, the lack of evidence of such a history is conspicuous both in the New Zealand flora and vegetation and in the literature that describes them.

Speculation about the natural role of fire on New Zealand indigenous vegetation gained momentum in the 1950s, when debate centred about the relative effects of fire and climate on the post-glacial distribution of grassland and forest (e.g., Holloway 1954). More recent palynological evidence has demonstrated that fire has affected the vegetation of at least inland South Island for several thousand years (e.g., McGlone 1988).

Mark (1955) showed evidence of former forest on a montane area supporting tussock grassland in Otago, and suggested that anthropogenic fire caused the change. Cumberland (1962) attributed the recent destruction of New Zealand forests to widespread use of fire by Polynesian settlers from about 1000 AD, and Molloy et al. (1963) provided radiocarbon dating of logs and charcoals to support this chronology. Early European explorers in New Zealand reported forest fires lit by the Polynesian inhabitants as late as the mid-19th century (e.g., Tuckett 1857, in Hamel 1977), and the Europeans themselves burnt the vegetation from the time of their first arrival.

Concerns about the effects of fire on the native vegetation, particularly when associated with grazing by introduced mammals, began to appear soon after European agriculture was established in New Zealand (e.g., Buchanan 1868; references in Mather 1982). However, most of the subsequent literature focused on agricultural development of burned and degraded grasslands by the use of introduced forage species and the application of fertiliser. Little research was carried out on the effects of fire on the native flora or vegetation until the mid-1960s. For the next 15 years, most studies which included the effects of burning involved only the dominant species of southern South Island montane tussock grasslands, *Chionochloa rigida* (e.g., O'Connor & Powell 1963, Mark 1965a,b,c, 1969, Connor 1966, Rowley 1970, Payton & Brasch 1978, Payton & Mark 1979). More recently, while research on the effects of fire on *Chionochloa* continued, interest in other components of grasslands and other vegetation types increased (e.g., Mark et al. 1979, Allen & Partridge 1988, Enright 1989, Timmins 1992). Nevertheless, despite the production of two major reviews addressing the role of fire in tussock grasslands (McKendry & O'Connor 1990, Basher et al. 1990), publications on fire in New Zealand ecosystems are still

rare, and the use of prescribed fire in New Zealand reserves even rarer (e.g., Calder et al. 1992).

In contrast, the uses and effects of natural and prescribed fire on vegetation have been described for a variety of plant communities world-wide, especially in North American (e.g., Wein & MacLean 1983) and Australian (e.g., Gill et al. 1981) forests, and the Mediterranean climate ecosystems of Australia, Europe, North America and South Africa (e.g., Mooney & Conrad 1977, Conrad & Oechel 1982).

Although there is a wealth of information on the effects of fire on grasslands, shrublands and forests, there have been few studies of fire in wetlands and little synthesis of the sparse information available (Kirby et al. 1988). In North America, fire has been used to control succession and weeds in coastal and inland marshes, but evaluation of fire as a management tool for wetlands has not been rigorous; wetlands management as a whole is poorly based on theory and the results of experiment and monitoring. In North America, although fire in deep peat soils has received attention, only recently have studies begun to address nutrient release, mineral cycling and other chemical effects of fire on the soil and subsequent vegetative vigour and productivity (Kirby et al. 1988).

Ecosystems elsewhere that show adaptations to fire have no exact equivalents in New Zealand, but some general principles of their fire ecology can be applied to gain an understanding of how fire affects New Zealand shrublands, grasslands and wetlands.

Fire has not previously been used as a management tool for the conservation estate in New Zealand, and there is a variety of opinions about the desirability of using prescribed fire as a conservation management tool. Fire can be considered to have a potential role in conservation management for several reasons, for example:

- to prevent natural succession and maintain a desired vegetation type with specific conservation values;
- to maintain habitats for plant or animal species;
- to reduce fuel loads and minimise the possibility of accidental fires.

However, the use of fire for conservation management raises a number of issues that need to be addressed before an effective fire management policy can be implemented. These include:

- the lack of scientific knowledge of the effects of burning on species and communities, rates of regeneration following fire, and the effect of fire on non-target species;
- the necessity for fire to maintain conservation values in some communities;
- when and how often should prescribed burning be carried out;
- the need for adequate fire control;
- the potential for undesirable weeds to increase.

3. Objectives

The objectives of this study were to:

- provide a summary of the scope of previous scientific reviews of fire ecology in New Zealand;
- review the overseas literature on the use of fire for management of grasslands, shrublands and wetlands for conservation purposes;
- review techniques for monitoring fire impacts and recovery;
- make recommendations for use of fire in ecological management in New Zealand.

4. Summary of previous reviews of fire ecology and management in New Zealand

McKendry and O'Connor (1990) undertook a review of the ecology of tussock grasslands for the Department of Conservation. This was wide-ranging and covered the use of tussock grasslands for both production and conservation purposes, the impacts of weeds and pests, and management approaches for the conservation of tussock grasslands. It considered the management of protected scrub and wetlands in tussock grassland zones, outlined the scope of present monitoring of tussock grasslands, and suggested some priorities for future monitoring.

McKendry and O'Connor (1990) recommended that the objectives and methods for conservation management of tussock grasslands should be formulated with regard to their natural and cultural histories. They described and classified short and tall tussock grasslands according to the formative and successional determinants of each type, in particular the stress and disturbance regimes that characterise each and its environmental niche (climate, soils). The two main disturbance elements considered were fire and grazing.

McKendry and O'Connor (1990) suggested that the indigenous grasslands below the timberline are largely seral vegetation maintained by fire, which has recently been accompanied by grazing. These two disturbance factors prevent the grasslands reverting to shrublands and forest. The relevance of continental (North American, Australian and African) rangeland fire research and management was considered to be only indirect since fire was a recurrent and critical part of ecosystem evolution in continental rangelands. While acknowledging that fire was infrequent in natural New Zealand landscapes, McKendry and O'Connor (1990) suggested that the seral nature of lowland and montane tussock grasslands provides some relevance for overseas research and management strategies. They briefly outlined some areas where prescribed burning is a commonly used tool (for both production and conservation) and the reasons for this, fire suppression issues, and the impact of fire on tall

tussock and short tussock grasslands and shrubs in New Zealand. They suggested that decisions concerning the requirement for or significance of fire in protected tussock grasslands depend on an understanding of fire impacts and successional processes, including the life strategies of individual species.

The Basher *et al.* (1990) review was primarily aimed at assessing the role of fire in the management of grasslands and shrublands for production purposes, and the ecological implications for the sustainability of production under the current fire management regimes. It briefly described the history of fire in the New Zealand landscape, the above- and below-ground heating regimes in grassland and scrub fires, species adaptations to fire, plant community successional responses to fire, and the impact of burning on soil physical, chemical and biological properties. Many long-term effects of fire are unknown, but there is a widely-held view that in New Zealand the widespread and repeated historical use of fire, in association with heavy grazing and limited application of fertiliser, has led to degradation of vegetation and soils on extensively managed rangeland.

There are many data on the effects of burning on snow tussocks (*Chionochloa* species) but little scientific information on the effects of burning on other plant communities, on soils, or on fauna. Few native plants display specific adaptations to fire. However, many (particularly herbaceous) species may survive fire through resprouting, enhanced flowering and seed germination. Snow tussocks show an initial leaf elongation, tillering and flowering response to burning but total biomass takes more than 15 years to recover. Spring burning has the least detrimental effect with late summer-autumn burning resulting in greater tiller mortality and slower recovery. Burning in combination with grazing can be effective in suppressing shrubs within grassland.

Fires in grassland and grassland-scrub typically result in a relatively low temperature rise, short duration of temperature rise, and are characterised by a rapid decrease in temperature below the soil surface. Soil temperatures rarely exceed 100°C in grassland fires and 200°C in scrub fires. The direct effects of soil heating are spatially variable due to variation in heating, but are generally confined to the top 10 to 20 mm of the soil where part of the litter layer, soil organic matter and soil and litter dwelling organisms may be destroyed. There is potential for the loss of organic matter and plant nutrients, by volatilisation, leaching and erosion, but no direct experimental data are available for New Zealand on the magnitude of these effects. Changes to soil microorganisms are likely to be especially important in view of their role in transforming plant residues and nutrients in soils and their sensitivity to an interruption in food supply induced by burning. There is generally little direct effect on soil physical properties unless heating has been particularly severe, although development of water repellency is likely. Loss of plant cover increases the potential for wind and sheet erosion and movement of soil by needle ice.

Both O'Connor and McKendry (1990) and Basher *et al.* (1990) suggested fire management policies should be based on:

- a clear definition of the objectives of burning;
- an understanding of the methods by which those objectives may be achieved;
- an understanding of the short-term and long-term consequences of burning on vegetation and other ecosystem components (e.g., fauna, soils).

5. Fire ecology of grasslands, shrublands and wetlands

5.1 HISTORICAL EVIDENCE OF FIRE FREQUENCY IN NEW ZEALAND ECOSYSTEMS

Evidence for long-term historical fire impacts and fire frequency comes from two sources:

- studies of vegetation history derived from pollen analysis that provide a picture of regional and local vegetation response to fire through changes in the proportions of arboreal and non-arboreal pollen and their association with variation in charcoal abundance;
- studies of subfossil forest remains (on the surface or in sedimentary sequences) that provide direct evidence for local forest destruction by fire.

Fire has been a feature of the New Zealand environment for many millions of years (Stevens et al. 1988), but before the arrival of humans, fire is considered to have been too infrequent to be of major ecological importance (Wardle 1984) since the New Zealand flora shows a low degree of specific adaptation to fire (Basher et al. 1990). Fire has been a dominating influence on vegetation patterns during the last 1000 years with Polynesian and European use of fire a major factor determining the extent of contemporary indigenous grassland below timberline in lowland and montane areas of both the eastern South Island (Molloy et al. 1963, McGlone 1983, 1989) and the central North Island (McGlone 1983, Rogers 1994).

Molloy et al. (1963) first described the distribution of radiocarbon dated subfossil forest remains in the eastern South Island. They showed that forests were once extensive throughout the region but had been removed by fire, especially during the Polynesian occupation of New Zealand. Twenty four of thirty one radiocarbon dated samples yielded dates less than 1000 yr BP. A small number of samples record earlier natural fires dating back to c.6500 yr BP. All the dated samples were of tree or shrub species (mainly *Podocarpus*, *Nothofagus*, and *Leptospermum*), while the present vegetation on the sites was primarily tussock grassland (both snow tussock and fescue tussock, *Festuca novae-zelandiae*) and bracken fern (*Pteridium esculentum*) and other shrub species. Molloy et al. (1963) mapped the distribution of charcoal and showed that it is commonly found throughout the eastern South Island from Marlborough to Southland.

A more comprehensive analysis of the temporal distribution of radiocarbon dated charcoals has recently been carried out by Basher et al. (in prep.) that confirms the interpretation of widespread forest destruction during the last thousand years and provides data on the frequency of earlier natural fires. All the dated charcoals were from the eastern South Island where relatively low rainfall, marked seasonal soil water deficits, periodic longer-term drought, and seasonal strong hot fohn winds are characteristic. About half (86 of 169 dated samples) are younger than 1000 yr BP, with the oldest samples beyond the

range of radiocarbon dating (>50,000 yr BP). Basher et al. (in prep.) estimate variation in rates of charcoal production through the Holocene from analysis of the number of dated charcoals per unit time. During the Holocene, rates of charcoal production increase from 5.2 events per thousand years between 3500 and 10,200 yr BP, to 143 per thousand years between 345 and 700 yr BP. Changes in rates of charcoal production occur at c.250, 700, 1000, 2900, and 10,200 yr BP. The two oldest changes in rate are related to the variation in the frequency of natural fires and probably reflect climatic variation. The data suggest that natural deforestation by fire had increased from about 2900 yr BP and that rates of charcoal production increased dramatically (by an order of magnitude) with the arrival of Polynesians in New Zealand. Dated charcoals and deforestation reach a peak between 700 and 500 yr BP with widespread forest destruction throughout the eastern South Island. Analysis of multiple charcoal layers dated at individual sites, and regional patterns of dated charcoals suggests intervals between natural fires are between 500 and 2000 yrs.

No similar analysis of subfossil charcoal material has been carried out for the North Island but McGlone (1983, 1989) and Rogers (1994) suggest a similar record of events is evident with the beginnings of deforestation by c.2500 yr BP and an acceleration after 1000 yr BP.

The pollen evidence from throughout New Zealand confirms this sequence of events and provides evidence of vegetation response following fire, with implications for the inferred frequency of burning. Many of the pollen diagrams record early deforestation between 3000 and 2000 yr BP (McGlone 1973, Enright et al. 1988, Rogers & McGlone 1989, McGlone et al. 1995). This period is characterised by a decrease in pollen of tree species and a rise in grass species associated with increased charcoal percentages, followed by a recovery in tree pollen.

Forest destruction during the Polynesian era is recorded in pollen diagrams from Northland (Enright et al. 1988), Waikato (Newnham et al. 1989), Bay of Plenty (McGlone 1983), Taranaki (McGlone 1983), Wanganui (Bussell 1988), central North Island (McGlone 1983), Hawkes Bay (McGlone 1978), Marlborough (McGlone & Basher 1995), Canterbury (McGlone 1983), Otago (McGlone et al. 1995), and Southland (McGlone & Bathgate 1983). Record locations range from coastal sites to high altitude wetlands, and indicate a widespread replacement of woody vegetation by grassland and fernland at low altitude and montane sites. A significant increase in bracken fern spores is recorded in association with the increase in grass pollen and charcoal fragments and the decrease in tree pollen in response to Polynesian deforestation (McGlone 1983, Enright et al. 1988, McGlone et al. 1995). This is interpreted to indicate that vegetation was burnt frequently over wide areas to maintain the bracken fern that is represented so strongly in most pollen diagrams. McGlone (1983) suggested that bracken is normally rapidly suppressed by shrubland and forest regeneration, and would only be maintained by frequent (perhaps every three years) burning. This contrasts with the records of earlier natural fires in which bracken barely features in the pollen diagrams, and which are interpreted to indicate fire destroyed forest over large areas but with long intervals between fires. Studies of succession in reserves in Otago (Calder et al. 1992) and central North Island (Rogers & Leathwick 1994) illustrate how

rapidly succession to shrubland occurs within tall tussock grasslands. In both cases, only a few decades were required for the transformation. These results also suggest a high frequency of burning through the Polynesian era to maintain grassland and fernland.

Changes in stratigraphy and vegetation composition at some of the wetland sites suggest that their present characteristics are fire-induced. The change from forest to grassland and fernland has altered water balance, and possibly nutrient input, sufficiently to affect the nature of some wetlands directly. For example, at Ponaki wetland in Northland the development of raupo wetland is considered to be a result of fire-induced erosion and sedimentation affecting drainage of, and nutrient supply to, the valley floor, forming a wetland where none had existed previously (Enright et al. 1988). Similarly, at Nokomai in central Otago rapid Sphagnum growth since 600 yr BP is inferred to be a response to increased runoff resulting from the burning of vegetation above treeline (McGlone et al. 1995).

The pollen and subfossil evidence unequivocally indicates that much of the indigenous vegetation below treeline, and some of that above tree line, has been fire-induced and maintained over the last 1000 years when fire frequency has been higher than at any other time throughout the Holocene. Before 3000 yr BP, 85 to 90% of New Zealand was covered in forest, with non-forest communities restricted to alpine areas, disturbed sites (e.g., floodplains, recent landslides, active sand dunes), frost-prone valley floors, very stony or infertile soils, and some wetlands (McGlone 1989). By 1000 yr BP, an unknown area had been deforested by natural fires, but by the time of European settlement half of the original forest cover had been removed (Masters et al. 1957). Thus, in the absence of management to maintain them in their present state, the grassland and shrubland communities within reserves below tree line will continually be transformed through successional processes. Williams et al. (1990) drew a similar conclusion for pakihi wetlands from their studies on the West Coast of the South Island.

5.2 RESPONSE TO FIRE

Basher et al. (1990) noted that burning impacts vary according to the intensity and temperature of fire, the vertical distribution of heat, and fire frequency and seasonality. These parameters depend on the type, amount and disposition of fuel, weather conditions, soil moisture, wind, time of day and year, and post-fire weather and management. The temporal and spatial variability of fire intensity and duration result in patchy patterns of combustion, survival and recovery of vegetation, and heating of soil.

5.2.1 *Frequency and season of fire*

General

Kilgore (1983) defined a natural fire as one that burns within the range of fire intensities, frequencies, seasons and sizes, and that yields the range of fire effects, found in that ecosystem before the arrival of technological Europeans.

He considered that, in fire-adapted vegetation, if a fire meets these criteria then its method of ignition is irrelevant.

Because increasing fuel accumulation with time leads to more intense fires, fire frequencies and intensities are closely related. Both follow a negative exponential model that applies to the age distribution of fire-controlled vegetation, in which about two-thirds of the stands will be younger than the average fire cycle and one-third older (Rowe 1983).

Fire frequency has a major effect on the balance between species that regenerate vegetatively when burnt (sprouters) and those that regenerate from seed (seeders). If fires are too frequent, seeders tend to be eliminated because they seldom reach maturity, whereas propagation of sprouters is stimulated (Kruger 1983). Sprouters are also favoured by more humid environments, where their canopy cover increases rapidly after fire, excluding seeders (Specht 1981). Seeders may be favoured by a lengthy interval between fires which allows greater thinning of sprouters and greater fuel accumulation, and thus more and larger post-fire openings in the vegetation, as well as less competition after fire (Keeley 1977).

Fires tend to occur in the summer half-year of the mediterranean climate, but times of high fire risk are more widely spread. They are strongly controlled by current weather and only partly correlated with season (McCutcheon 1977). The most important vegetation characteristic in determining fire intensity is the amount of dead material (Rogers 1982).

Fire frequency in mediterranean climate shrublands is very variable, but relatively few areas experience fires at intervals of less than 5 or greater than 20 years (references in Kruger 1983). The lower limit represents the minimum time required to build up sufficient fuel to carry fire, and is determined by vegetation structure and productivity. The upper limit reflects the frequency of chance ignition events. Where the interval between fires is unnaturally short, for example in frequently-burnt *Quercus coccifera* garrigue, tufted grasses and other perennial herbs become prominent (Trabaud 1973, 1982).

The seasonal effects of fire in savanna grassland of South Africa are related to the physiological state of grasses: actively growing plants (late spring and autumn) are more susceptible to damage from fire than dormant plants (winter and drought-induced summer dormancy) (Trollope 1984). Season of fire also affected seed production by grasses in this ecosystem.

New Zealand

In New Zealand, burning for agricultural management is considered less damaging in early spring, before new growth occurs and while the vegetation is relatively moist, than in other seasons (Basher et al. 1990). This contention is supported by some research findings on *Chionochloa rigida* grassland (e.g., Mark 1965c, Allen & Partridge 1988, Calder et al. 1992) and short tussock grassland (e.g., Sewell 1948), but has not been tested elsewhere. Similarly, some effects of fire frequency, particularly when linked with grazing, are documented for *C. rigida* (e.g., Gitay et al. 1992), but not for other species and vegetation.

5.2.2 *Plant species responses*

General

Species responses to fires depend on the life cycles of plants, the fire regimes to which they are subject, and the local post-fire environment (Gill 1981). To survive in an environment subject to burning, a plant species not only has to survive a single fire, but also to persist through a fire regime defined by the frequency, intensity and season of burning (Harrington et al. 1984). The above-ground parts of most shrub species are killed during most shrubland fires, although only 60-80% of above-ground biomass may be consumed (Christensen 1985). The effect of fire on plants is modified by their phenologies; for example, early-sprouting grasses are more sensitive to spring fires than those which sprout in summer (Rowe 1983). At a longer time scale, species that require several years to reach reproductive maturity are more sensitive to short fire cycles than those which mature rapidly (Gill 1981).

Rowe (1983) proposed modes of persistence based on two forms of post-fire reproduction. Disseminule-based persistence (seeders or non-sprouters; Noble & Slatyer 1977) includes invaders (highly dispersive, shade-intolerant pioneering fugitives with short-lived seeds), evaders (species with long-lived seeds stored in soil or canopy) and avoiders (shade-tolerant species that slowly reinvade burnt areas; late successional). Vegetative-based persistence (sprouters; Noble & Slatyer 1977) includes resisters (shade intolerant species whose adult stages can survive low-intensity fires), and endurers (resprouting species, shade tolerant or intolerant, with buried perennating buds).

Common features which adapt plant species to tolerate fire (Trabaud 1982, Harrington et al. 1984) include aerial regenerative buds protected by thick bark; growing points protected by soil or leaf sheaths; buds on subterranean structures such as lignotubers and rhizomes; root suckers, corms and bulbs; dormant buried seeds which germinate in response to fire; and dormant seeds which are released from storage on the plant and stimulated to germinate by fire. Gill (1977, 1981) pointed out that such adaptations may be adaptive to variables other than fire; drought and grazing are obvious examples. The relative investment of shrub species in seed production versus vegetative structures depends on the fire return interval, pre-fire resource limitations, fire intensity and associated shrub mortality, and postfire seedling demography (Christensen 1985).

In mediterranean climate ecosystems, species that reproduce from seed rely on dormant seed storage either in the soil (the norm for chaparral species; Hanes 1977) or in protective organs on the parent plant (serotiny; many shrubs of South African and Australian heathlands; Gill & Groves 1981) until the organ or parent plant dies, usually as a result of fire. In both strategies, fire removes competing vegetation, leaves a suitable seedbed, and stimulates germination. After fire, increased insolation can result in soil heating, air temperature above burnt areas is more variable than on unburnt areas, and available light is increased on burnt areas. Such microclimate changes can lead to temperatures sufficient to break the dormancy of buried seeds, to light levels that greatly influence seedling success, and to other effects (Christensen 1985).

Several Australian monocot species are stimulated to flower by burning, although the causes of the flowering response vary widely, and include leaf removal, increased nutrient availability from ash, increased light, decreased competition, and changes in the diurnal temperature regime (Gill 1981).

Large crops of seed able to take advantage of post-fire conditions can be produced by massive post-fire flowering, by retention of seed produced from several inter-fire flowerings to be released by the stimulus of fire, or by accumulation of dormant seeds in the soil which are stimulated to germinate by fire (Gill 1981). Germination of dormant seeds in the soil seed bank after fire is not widespread or uniform and is confined to areas of high fire intensity and high surface seed density. Most seedlings subsequently die because the chance of rainfall suitable for establishment is low (Hodgkinson & Harrington 1985).

In species that recover from fire by sprouting from buried buds, the result may be either no population increase, where the regrowth occurs only from original pre-fire apices, or population increase, where outgrowth of root buds or rhizomatous buds occurs (Gill 1981). The vitality of sprouting response may be reduced substantially by repeated burning.

Fire intensity measured above ground may bear little relation to the survival of underground perennating organs, which is controlled by the depth to which humus is burnt and the temperature profile in the soil (Rowe 1983).

New Zealand

Both seeding and sprouting strategies are seen in the few New Zealand plant species which have morphological or physiological adaptation to fire.

A few species show enhanced seeding and germination after fire; e.g., *Chionochloa rigida* (a physiological response in the first season after burning; Rowley 1970), manuka (*Leptospermum scoparium*; release of seeds from serotinous capsules immediately after fire; Burrell 1965), *Hebe* and *Gonocarpus* (Basher et al. 1990). However, apart from the tree kowhai (*Sophora* species) and possibly *Carmichaelia* shrub species, there is little evidence of hard-seededness in the native flora, so native grasslands and shrublands lack a buried seed bank of long-lived seeds (Basher et al. 1990).

Sprouting is the more common strategy. Some protection is afforded growing points close to the ground by thick and densely packed leaf sheaths in species of *Celmisia*, *Aciphylla*, *Phormium*, grasses and sedges (Basher et al. 1990). Tussock grasses, bracken fern, and the shrubs *Cassinia* (Wardle 1991) and matagouri (*Discaria toumatou*; Daly 1969) may resprout from buried rhizomes or buds following fire, as do some ferns, Liliaceae and Orchidaceae.

The effects of fire are best documented for *Chionochloa* species. O'Connor (1963), O'Connor & Powell (1963), Mark (1965c), Rowley (1970) and others have determined the recovery rates of *C. rigida* and *C. macra* after fire, and demonstrated that recovery is most rapid in the absence of grazing. Changes in nutrient regimes within the plant in response to fire are also well described (e.g., Williams & Meurk 1977, Payton et al. 1986). In summary, it takes at least 14 years for *Chionochloa rigida* plants to recover their pre-burn state as measured by nutrient concentrations and distribution, leaf elongation, and flowering intensity, and more than 15 years to recover the biomass of unburnt

plants (Basher et al. 1990). Recent studies suggest that at least 25, and up to 50 years are required for burnt *C. rigida* plants to regain the stature of unburnt plants in a subalpine environment (K. Jehn, Landcare Research, unpub. data).

5.2.3 *Plant community responses*

General

The effects of fire depend on the interaction of several variables (Gill & Groves 1981): temperature (determining nutrient losses and survival of perennating organs; Evans & Allen 1971), intensity (a synthesis of temperature and rate of spread), frequency (the major variable affecting species composition), grazing, post-fire management (Basher et al. 1990), and season.

Even in areas where there is a long history of vegetation management by fire, it has not been possible to develop reliable models of plant community behaviour following different combinations of fire, rainfall and grazing (Noble et al. 1986). Fire and grazing regimes both favour plants which have protected growing points or are able to resprout after defoliation. However, fire is indiscriminate, whereas grazing animals mainly consume living plant material and are selective between plant species. Fire is an irregular and acute occurrence, whereas grazing is regular and chronic (Harrington et al. 1984). Heavy browsing pressure after a fire may increase plant mortality from a second fire, especially if it occurs within a few years of the first (Gill & Groves 1981). Grazing can both affect and interact with the fire regime by reducing the quantity of fuel and by changing plant species composition (Noble et al. 1986).

Main (1981) contrasted the effects of burn size: small burns are rapidly reoccupied by invertebrates and intensely grazed by herbivores which survive in adjacent unburnt areas, favouring unpalatable plant species, whereas large burns are less affected or unaffected by grazing and are only slowly reinvaded by invertebrates because shelter for these animals is distant. Frequent large burns favour fast-reproducing weedy plant species, and infrequent large burns maintain plant species diversity.

Overseas shrublands

Shrublands in which fire is a dominant ecological factor include the ericaceous heathlands of north-west Europe, the fynbos of South Africa, heaths of southern and eastern Australia, garrigue and maquis of the Mediterranean, and the chaparral of western USA.

Large scale fires in these shrublands generally follow seasons of above-average rainfall, when there has been an unusually high production of plant biomass and hence of fuel. Rainfall patterns following fire dictate what species will germinate and establish successfully, and thus also govern the amount and type of herbivory, with consequent effects on species composition and fuel buildup. Grazing reduces the competitive stress imposed by herbaceous perennials, including pasture plants, on shrub seedlings, and allows the latter to grow more or less unchecked, favouring the establishment of shrub dominance (Hodgkinson & Harrington 1985).

Shrublands suffer either surface or crown fires. In semi-arid shrublands such as fynbos, foliage is very flammable because of the presence of terpenes, and crown fires readily develop (Trollope 1984). In more mesic ecosystems, such as subtropical savanna, shrub foliage is relatively non-flammable. Crown fires are relatively rare, and develop only under extremely dry conditions. Fire tends not to penetrate far into stands of shrubs because of both the relatively moist conditions under the canopy and the lack of fuel from a herbaceous understorey, so as stands increase in extent, fire becomes a less effective management tool (Trollope 1984).

Christensen (1985) noted several effects of fire in shrublands. Despite nutrient losses from smoke and runoff, nutrient availability is usually higher and more variable immediately after fire. Reduced leaf area reduces evapotranspiration and thus increases water availability, even though increased surface heating may reduce available water in surface horizons. Allelochemicals are removed by destruction of their source plants and may be denatured in the soil. Invertebrate and vertebrate herbivore populations change markedly.

Kruger (1983) summarised the characteristics of post-fire succession in mediterranean-type ecosystems (Table 1). After fire in mediterranean climate shrublands, the species that were present before the fire usually regenerate and develop through their life cycles, so the vegetation regains its pre-fire flora and structure (Specht et al. 1958, Hanes 1971, Russell & Parsons 1978, Trabaud 1981). However, the time required for recovery is variable, and floristic composition can change markedly through that period, and between different fire events, depending on the regeneration strategies of the flora, the season and intensity of the fire, the interval between fires, and weather subsequent to fire (Kruger 1983 and references therein). The main feature of floristic change is the appearance soon after fire of "fire ephemerals" and other species which depend on fire for regeneration, and their subsequent disappearance as longer-lived overstorey species gradually suppress them (Specht et al. 1958, Hanes 1977).

In mediterranean climate ecosystems, prolonged absence of fire can have two effects (Kruger 1983). Australian heath and South African fynbos show cyclic succession where shrubland species alternate in dominance or regenerate into gaps caused by senescence. Alternatively, as seen in Californian chaparral and some Australian heaths, senescence of dominant shrub species leads to prominence of less abundant species already present, including grasses, or may allow invasion by species of an alternative vegetation type, such as forest.

The conditions that permit invasion or re-invasion of indigenous elements into South African fynbos after fire also permit invasion by some introduced tree and shrub species. The latter behave like the indigenous invaders, but excel because they have better dispersal abilities, as well as greater fire resilience which is conferred by higher growth rates, greater stature, serotiny, and copious seed production (Richardson & Cowling 1993).

In an environment more akin to those found in New Zealand, Kirkpatrick & Dickinson (1984) described five classes of species recovery from fire in Tasmanian alpine shrublands: no recovery (adults and seeds killed by fire); regeneration from seed but failure to regain pre-fire cover; vegetative recovery but failure to regain pre-fire cover; regeneration from seed to increase cover

TABLE 1. CHARACTERISTICS OF POST-FIRE SUCCESSION IN MEDITERRANEAN-TYPE ECOSYSTEMS. FROM KRUGER (1983).

	CHAPARRAL	MATORRAL	MAQUIS/GARRIGUE	AUSTRALIAN HEATH	FYNBOS
annuals + short-lived fire ephemerals	dominant post fire	none	none?	rare	rare
immigrant post-fire annuals	none?	none?	often abundant	none?	none?
geophytes	some	?	many	prominent	prominent
persistent herbaceous perennials	very few	abundant; decreasing	grasses stimulated; rare in old stands	sedges, restiads dominant; remain prominent	grasses, sedges, restiads dominant; remain prominent
shrub regeneration mode	about 50% resprout	most resprout; few by seed only	most resprout; few by seed only	both modes prominent	both modes prominent
timing of recruitment	immediately post-fire	?	immediately post-fire	immediately post-fire	immediately post-fire
fire-stimulated shrub germination	yes	no	no	yes	yes
redevelopment of canopy cover	100% in 10-20 yr	?	closed canopy in 5 yr	100% in 10 yr	70-90% in 2 yr
species richness (S) trends	S decreases with age	?	S decreases with age	S decreases with age	S decreases with age; sometimes an initial increase
effect of long absence of fire	stagnation	?	?	stagnation or trend to woodland	stagnation or immigration of evergreen forest species

post-fire; vegetative recovery to increase cover post-fire. Few alpine species were unable either to recover vegetatively, or from seeds stored in the soil, or from seed dispersal from unburned areas. However, species lacking all these attributes are major dominants in unburnt areas of coniferous and deciduous heath, so fire results in dramatic changes in dominance and structure, eliminating much of the endemic shrub element and favouring more widespread species.

New Zealand shrublands

Natural shrublands in New Zealand are usually confined to environments where climatic (e.g., the subalpine zone) or edaphic (e.g., ultramafic soils) constraints prevent succession to forest, to primary succession such as that following volcanism or glacial retreat, or to sites where succession is continually curtailed by events such as landslides or shifts in river courses. Wardle (1991) described natural shrubland dominated by stunted, slow growing, small-leaved trees and shrubs, on thin, leached soils over siliceous and ultramafic rocks and on gley podzols on fluvio-glacial terraces in the west of South Island and on Stewart Island. Dwarf podocarps, manuka and epacrids are prominent. Similar vegetation occurs in the subalpine zone of North and South Islands, differing mainly in its prominence of broad-leaved woody composites such as *Olearia colensoi*, and *Dracophyllum* species (Wardle 1991). Shrub-heaths dominated by *Dracophyllum* species, filiramulate shrubs, manuka and several sub-shrub epacrids, accompanied by small podocarps, flax (*Phormium tenax*), several rush and sedge species, ferns, mosses and lichens, occur on frost flats and mires of the central North Island volcanic plateau. Manuka-dominated heathlands occupy infertile soils with perched water tables ("gumlands" and similar environments) of the Coromandel Peninsula and Northland (Wardle 1991, Enright 1989). Kanuka (*Kunzea ericoides*) and manuka form open heathland on very dry sites, where succession to forest is extremely slow or non-existent. Kanuka-dominant shrubland was extensive on well-drained soils of the Canterbury Plains in post-glacial times (Molloy & Ives 1972).

The only substantially fire-adapted New Zealand shrublands are manuka-dominated gumland heathland, which has a fire history of at least 16,000 years, and Canterbury kanuka shrubland where soil charcoals indicate a history of fire throughout the post-glacial era (Wardle 1991). Nevertheless, both are perpetuated by edaphic factors (soil nutrients/waterlogging and soil moisture deficit, respectively) as much as fire. No other natural shrublands show fire adaptation in the forms found in mediterranean climate shrublands, although a few shrub species with fire adaptations may be present.

The extent of natural New Zealand temperate shrublands has been substantially reduced by Polynesian and European fires and agricultural development. Most of those which remain have been burnt, and many thus converted to shrubland dominated by manuka, or to grassland (Wardle 1991). Where seed of exotic tree and shrub species such as lodgepole pine (*Pinus contorta*), gorse (*Ulex europaeus*), heather (*Calluna vulgaris*), sweet brier (*Rosa rubiginosa*) or *Hakea* is available, repeated burning often results in secondary vegetation dominated by these, which are all well-adapted to fire. In the absence of these exotics, secondary manuka and kanuka shrublands are found where forest has been

destroyed by fire throughout New Zealand. They are maintained by fire, harsh environments, distance from sources of forest seed, and browsing of palatable seedlings of forest species by introduced animals (Wardle 1991), although kanuka can be eliminated by frequent burning in some places (Williams 1983).

Fire-induced shrublands are widespread over a wide altitudinal range of drier eastern South Island areas, where they were recorded by early European explorers and naturalists (references in Cockayne 1958) and can be attributed largely to the destruction of forests by 1000 years of anthropogenic fire (Wardle 1991). At higher altitudes these are characterised by the dominance of *Dracophyllum* species, particularly *D. uniflorum*, an abundance of seral and filiramulate shrubs, especially *Hebe*, *Cassinia* and *Coprosma* and, where moisture permits, the presence of fire-resistant mountain flax (*Phormium cookianum*). At lower altitudes, manuka or matagouri (*Discaria toumatou*) can be dominant (Wardle 1991), the latter often accompanied by *Melicactus alpinus*, *Coprosma*, *Carmichaelia* and small-leaved deciduous *Olearia* species in semi-arid areas, where Wardle (1991) considers such vegetation may be an edaphic climax.

Although Wardle (1991) considered these shrubland types to be fire tolerant, apart from serotinous manuka, the only shrub species in which adaptation to withstand fire has been recognised are *Cassinia* and matagouri, both sprouters. Other species rely on effective dispersal of large numbers of viable seeds to maintain their presence after fire.

Bracken fernland, naturally confined to well-drained porous soils such as coastal and inland dunes, increased substantially in extent as a result of continual burning by early Polynesians during the past millennium (Wardle 1991). It produces a substantial layer of very flammable litter, but recovers vigorously from fire from its deep rhizomes. Farming practice is to burn bracken in spring and oversow with introduced pasture species, but success in eliminating bracken depends on the stock density which can be maintained to trample the new fern growth (Wardle 1991). In the absence of fire, bracken may be succeeded by plant species of adjacent shrubland or forest (Wardle 1991), although establishing seedlings are vulnerable to summer drought (Partridge 1989).

Overseas grasslands/rangelands

Grass fires have a narrow zone of flames advancing across finely divided and homogeneously dispersed fuel, which is more sensitive to weather and burns more rapidly and completely than woody aerial stems and foliage (Daubenmire 1968). Grassland fires are seldom hot enough to oxidise organic matter more than a few millimetres below the soil surface but, after fire, air and surface temperatures increase as a result of increased insolation, and subsequent microbial activity could reduce soil humus content, although this could be offset by decay of the underground parts of fire-killed plants (Daubenmire 1968). The increased temperatures may also result in earlier sprouting and flowering of grasses.

Fire commonly favours forbs over grasses in both annual and perennial grassland, and annual grasses can increase at the expense of perennials as a result of burning semi-arid grasslands, especially if accompanied by overgrazing.

In most North American and African mesic perennial grasslands, burning increases productivity for a few years after fire, but the opposite is true in many semi-arid grasslands because of the unreliability of rainfall. Fire also tends to increase floristic diversity in grassland, at least temporarily, by reducing competition and preparing a suitable seed bed for invaders (references in Daubenmire 1968).

Gill (1975) claimed that fire is a natural phenomenon of Australian rangelands, and that populations of all their plant species are capable of surviving certain fire regimes. Fire management on Australian pastoral land has four principal objectives: control of woody plants, stimulation of herbage growth, reduction of wildfire hazard, and prevention of fire in sensitive communities. In Australian conservation areas, the main use of fire is to maintain ecological diversity, particularly a mosaic of foraging and shelter habitats to favour higher populations of animals (Hodgkinson et al. 1984).

The effects of fire on Australian rangelands have been summarised by Harrington et al. (1984) as follows:

- fire requires and consumes dead plant material;
- fire scorches, kills, and may consume, green plant material;
- nutrients in plant material are returned to the soil or lost to the atmosphere;
- the soil surface is laid bare;
- perennial plants are reduced, allowing recruitment of new plants;
- seed may be destroyed, dispersed or scarified ready for germination;
- phytotoxic chemicals in plant leaves are destroyed.

A pulse of regrowth and recruitment follows the first effective rain because of low leaf biomass and thus low competition for moisture and nutrients. This slows as competition increases until, in the absence of another fire, the vegetation is dominated by those plants which are largest, longest-lived, or can regenerate despite the presence of established plants.

Frequent and moderately intense defoliation has been shown to be essential to the maintenance of a vigorous and dense grassland sward in humid montane regions of South Africa where the high rainfall and moderate temperatures are suited to forest (Tainton & Mentis 1984). Under these conditions woody plants are confined to fire refugia, and ecologically more advanced communities seldom become established. The season of burning in this environment greatly influences the composition of the resulting grassland.

In the more arid grassveld of South Africa, absence of fire for several years can lead to replacement of palatable by unpalatable grass species, whereas under annual burning, perennial species die out and cover declines (Tainton & Mentis 1984). As in other ecosystems, the effect of fire in veld grassland is greatly influenced by the time of burning. The best recovery of palatable species occurs when the vegetation is burnt prior to the rise of soil temperatures in spring, particularly if the soil is moist.

Recovery times for burnt grass-dominated prairies of North America depend on the season, but even after fire in a dry season, most grass species recover their pre-burn productivity after three years (Wright & Bailey 1982).

New Zealand grasslands

Two reviews of ecology and management of New Zealand tussock grasslands comprehensively cover the known effects of fire in the main New Zealand grassland ecosystems (McKendry & O'Connor 1990; Basher et al. 1990). In the absence of grazing, patterns of recovery are similar to those described for overseas grasslands: an initial pulse of vigorous regrowth in the surviving plants and an increase in floristic diversity, particularly in forbs, followed by a decline in growth rates and diversity as the canopy dominants increase in stature. Burning at too-frequent intervals, or followed by grazing, leads to a decline in vigour, and the eventual demise, of *Chionochloa* species and their replacement by more tolerant native (e.g., blue tussock *Poa colensoi*, hard tussock *Festuca novae-zelandiae*) or introduced (e.g., browntop *Agrostis capillaris*, sweet vernal *Anthoxanthum odoratum*) grass species, and several predominantly exotic geophytic forbs (e.g., catsear *Hypochoeris radicata*, mouse-eared hawkweed *Hieracium pilosella*). In the absence of fire, montane fire-induced grasslands are succeeded by shrubland and forest within decades if a seed source is sufficiently close (e.g., Connor 1965, O'Connor 1982, Calder et al. 1992, Rogers & Leathwick 1994).

Overseas wetlands

Most studies of the effects of fire in wetlands concern the management of forested swamps and coastal marshes in north America, where timber production, availability of forage for mammals, and the maintenance of habitat for hunted waterfowl are important. For example, Anderson & Best (1982) noted that periodic fires are a major factor influencing dynamics of succession in cypress swamp. *Sphagnum* dominates recovery of the herb layer, most shrub stems are killed, but a significant number of trees resprout from stump or bole.

Fewer studies address conservation of biodiversity or natural ecosystems. Christensen (1981) found that the balance between moist savanna, grass-sedge bog, shrub bog, white cypress bog and deciduous swamp forest in the southeastern USA is determined by the frequency and intensity of fires. Eleuterius & Jones (1969) noted that annual fires maintained greater plant diversity and productivity in wetlands of south-eastern USA than no burning for 3 years. However, frequent fires over two decades decreased floristic diversity and encouraged invasion of exotic species into Florida wetlands (Austin 1976). Wein & Bliss (1973) found that the rare fire in tundra tussock vegetation on wet peat substrate resulted in loss of few species, but increased surface temperatures by charring, leading to a deeper biologically active soil layer for several years. Annual production recovered after two years. Fire burns selectively along lichen- and litter-covered ridges and hummocks of Canadian raised peat mires, leaving moist hollows unburnt. It can result in replacement of lichens by *Sphagnum*, and reduce cover of vascular plants for up to 20 years (Foster 1984, Foster & Glaser 1986).

One of the rare publications concerning fire and wetlands outside north America showed that in the vlei wetland communities of South Africa, fire increases soil exposure and promotes a less hydrophytic environment. Frequent fires in vlei wetlands result in successional change from reed swamp

communities to grassland, whereas a lack of fire results in establishment of hydrophytic woodland (Tainton & Mentis 1984).

Issues such as substrate stability and maintenance of water regimes have received some attention. Fires that remove organic substrate alter the elevation, hydroperiod, soil type and vegetation of wetlands (Duever et al. 1986). Charcoal layers in Sphagnum-dominated peatlands of western boreal Canada show that vegetation change, such as moss species composition, is limited to a few decades after fire, but peat accumulation rate declines with increasing fire frequency (Kuhry 1994). Fires in the swamp-marsh complex of Georgia, USA, affect peat development both by destroying peat and by changing the character of vegetation, e.g., by converting swamps to open marsh (Cohen 1974). Repeated fire impedes elevation of the marsh surface and succession to dryland vegetation by preventing accumulation of organic matter in North American coastal marshes (Daiber 1974).

There is comparatively limited information on wetland plant adaptations to, and requirements for, fire. Most plants of shrub bogs in the southeastern USA show adaptations to fire similar to those of dryland shrubland species, such as serotinous fruits and a capacity to sprout from subterranean organs (Christensen 1981). An example of species requiring fire to limit competition is that of pitcher plants and orchids in south-eastern USA, which require annual fire to maintain open habitat and are otherwise replaced with "sedge-woody" species (Eleuterius & Jones 1969, McDaniel 1971).

New Zealand wetlands

Wetland plants adapted to fire include flax, which resprouts from meristems protected by thick sheaths of leaf bases, rhizomatous sedges (e.g., *Carex* and *Baumea* species), and serotinous manuka (Wardle 1991). Other rhizomatous species, such as *Gleichenia*, have relatively shallow rhizomes and only survive cool fires that do not penetrate the peat surface (Clarkson & Stanway 1994). A major plant of infertile lowland to subalpine wetlands, wire rush (*Empodisma minus*) is killed by most fires (Wardle 1991, Clarkson & Stanway 1994). The liverwort *Marchantia berteroana* colonises wet heath and bog surfaces where mosses have been killed by fire (Wardle 1991).

There are few studies specifically of fire ecology in New Zealand wetlands, although several descriptions infer fire effects on vegetation patterns.

Shrubs, particularly manuka and small podocarps, can occupy much of the surface of South Island west coast infertile swamps and bogs, and these communities are fire-prone (Wardle 1991). Fire affects the proportions of woody and herbaceous species in such wetlands (e.g., Wardle 1977, Mark & Smith 1975), where it acts mainly to arrest succession to kahikatea (*Dacrycarpus dacrydioides*) or kahikatea-rimu (*Dacrydium cupressinum*) forest (Wardle 1977). The extent of fertile swamp is increased to the detriment of adjacent forest by fire favouring rhizomatous *Carex* species, particularly *C. coriacea*, which produce considerable fuel loads. Infertile swamps are less productive and less flammable, and the main effect of fire here is to kill resident manuka, *Dracophyllum* and *Coprosma* shrubs, which can soon re-establish (Wardle 1977).

Mires on low terrain near Lakes Te Anau and Manapouri have similar species to those of west coast wetlands, but are structurally similar to cushion bog, with a peat substrate, domed, *Sphagnum*-dominated, surface vegetation, and stands of manuka on drier substrates (Wardle 1991). Fire in a *Sphagnum*-wire rush bog near Te Anau reduced plant cover to less than 20% on drier areas, but this recovered to 90% in four and a half years, although vegetation composition still differed from the pre-burnt state (Timmins 1992). Drier parts of the wetland were most severely affected by fire, and their species diversity increased rapidly soon after fire as they were colonised by native and naturalised herbs and grasses. Most of these did not persist, and diversity steadily declined as the dominant native wetland plants increased their cover. Rhizomatous species, for example bracken, *Baumea tenax* and *Carex gaudichaudiana*, re-established most rapidly, and *Sphagnum* relatively slowly. Flax recovered steadily by resprouting, but the main woody species, manuka and inaka (*Dracophyllum longifolium*), regenerated relatively slowly from seed. There is some evidence that manuka regeneration is reduced as fire temperature increases. Inaka, other epacrids, and shrub podocarps, may take decades to recover (Timmins 1992).

Cushion bog is extensive on the Awarua Plain south-east of Invercargill, with sedges and flax on very wet ground and at stream margins (Kelly 1968). On drier ground bog merges into red tussock (*Chionochloa rubra*) grassland and inaka-manuka scrub. After fire in six vegetation types in this area, recovery followed a similar pattern to that observed at Te Anau. Rhizomatous species recovered most rapidly, and there was an initial increase in abundance of native and naturalised herbs and grasses which declined or disappeared after 15-22 months. Manuka re-established rapidly from seed, but re-establishment of epacrid shrubs was later and slower. Former dominants such as red tussock, wire rush, *Gleichenia dicarpa* and *Sphagnum cristatum* were slow to recover, the last by slow recolonisation of peat surfaces rather than regrowth of fire-damaged cushions. Cushion bog dominated by *Donatia novae-zelandiae* is particularly sensitive to, and slow to recover from, fire. The expansion of gorse on the bog was encouraged by fire (P.N. Johnson, unpub. data).

Wire rush, *Gleichenia*, *Baumea*, manuka and *Dracophyllum* are characteristic of natural and fire-induced pakihi communities in Westland and western Nelson, where drainage patterns and soil parent materials result in a mosaic of vegetation types dominated by different species (Williams et al. 1990, Wardle 1991). Most pakihi has been induced by burning former forest, and is maintained by repeated fires (Williams et al. 1990). In the absence of fire, pakihi vegetation is succeeded by forest except on soils incapable of supporting the latter. Williams et al. (1990) found that 12 m tall black beech (*Nothofagus solandri*), with an understorey of shrub podocarps, was established in a moribund stand of manuka at 63 years of age on gley podzol soil, but that up to 50 years were required for development of dense manuka stands on deep, wet, organic soils. As on other wetlands, introduced species are most frequent on disturbed sites, including those recently burnt (Williams et al. 1990). Many small plants characteristic of pakihi, including several orchids, are only common in low, open, herbaceous vegetation.

Miller (1993) outlined the implications of fire as a management tool in pakihi, and concluded that there are no ecological imperatives and few historical

guidelines for its use. He suggested that the adoption of a prescribed fire regime for pakihi would depend on a decision whether to try to maintain an ecosystem in a static state, which he considered would eventually fail, or to allow natural processes to occur with occasional management initiatives to control undesirable elements.

Fire resulting from vulcanism, lightning or spontaneous combustion is a natural phenomenon on Waikato peat bogs, occurring at frequencies of one to several hundred years (references in Clarkson & Stanway 1994). Vegetation recovery following fires on the oligotrophic Whangamarino wetland in the Waikato (Clarkson & Stanway 1994) was similar to that described for Te Anau. Undecomposed surface litter provided fuel for the fire, which removed this and most standing live vegetation but only slightly charred the saturated peat surface. Rhizomatous species (*Baumea*, *Schoenus*, *Gleichenia*) resprouted and grew rapidly in the initial post-fire period, whereas species that were killed, for example wire rush and manuka, re-established more slowly from seed. Species diversity increased rapidly after fire, when creation of bare peat surfaces and increased fertility allowed establishment of small herbaceous species such as bladderwort (*Utricularia*), sundew (*Drosera*) and orchids, then declined again as the dominant species increased their cover. One month after fire, bare peat was still predominant, but by six months *Baumea* and *Schoenus* cover was substantial and seedlings of most other herbaceous species were established. Within two to four years many early colonisers, including most introduced species, had died out. Cover of *Schoenus* and *Gleichenia* declined as that of *Baumea* and manuka increased to their maximum by six years, when the vegetation had recovered its pre-fire character. In the absence of fire, Clarkson & Stanway (1994) considered that succession would proceed to an increasingly species-poor and native-dominated flora as the peat surface rises and becomes increasingly dependent on rainfall for nutrient supply, as is seen on more advanced high moor peat domes at Kopuatai, Torehape and Moanatuatua. Clarkson & Stanway (1994) recommended that fires should be controlled to allow this succession to continue except where maintenance of large populations of early successional species was a priority. They considered that there should be wide consultation and strict controls before any burning was implemented.

5.2.4 Faunal responses

Shrublands and grasslands

There has been little research on the ways animal populations persist in fire-prone habitats (Main 1981). Fires moving downwind or downhill leave many areas unburnt, whereas those burning against the wind or uphill are more destructive and consume more of the fuel bed. The extent of a fire depends on topography, air currents, temperature and fuel, which result in a mosaic of vegetation patches important for the survival of animals. The patches become centres of refuge and shelter for vertebrates and invertebrates from which the surviving populations can reinvade the surrounding areas when the habitat has become suitable (Main 1981). Recovery of small mammal and bird populations after fire is most rapid where significant cover remains as a result of unburnt

patches of vegetation or the presence of rocky terrain (Tainton & Mentis 1984). Observations of animal behaviour (Main 1981) suggest that birds flee in front of smoke and flames whereas small mammals and reptiles seek refuge in burrows. Non-flying invertebrates can escape fire in the shelter of green growth or deep damp litter, but flying insects are frequently killed. After fire, herbivores suffer food deprivation until the vegetation regenerates. Exposure during this period, especially linked to extremes of heat, drought or cold, also increases the risk of mortality. Absence of shelter can also lead to increased predation. Frequent fires favour short-lived early-maturing animals, especially those with good dispersal.

Moreno & Villafuerte (1995) noted that a reduction in burning frequency of scrubland in a Spanish national park resulted in a decline in rabbit population. Experimental burning showed that rabbits favoured recently-burnt sites close to scrub cover.

In the South African grassland biome, greatest abundance and species richness of small mammals and birds occurs on areas never or rarely burnt, and up to three years after fire. This is interpreted by Tainton & Mentis (1984) as a bimodal distribution of these species into those adapted to fire climax and post-fire climax vegetation types, with relatively few species occupying the "moribund sward" which occurs between these two successional stages.

Warren et al. (1987) reviewed the effects of different types of fires on different classes of invertebrate pest. In general, pest control depended on the interactions of invertebrate life cycle, habitat preference, availability of refugia, and season and intensity of fire. Models to explain these were developed based on a four-phase sequence of impacts attributable to prescribed burning, which partition acute and chronic fire influences on arthropod populations: fuel development, combustion, shock, and ecosystem recovery (Warren et al. 1987). The size of the burned area affects the rate and extent of arthropod immigration from unburnt areas during the shock and recovery phases. Agile fliers and jumpers can escape most fires, but may fall victim to predators. Arthropods in the soil and those taking refuge in cracks, moist vegetation or under rocks, may escape. Post-burn effects include habitat loss, desiccation, increased predation, and emigration.

The effects of fire on insects in tropical African savannas include the disappearance of strong flying groups (dispersal or mortality), little effect on soil surface fauna (relatively low surface and sub-surface temperatures), and a decrease in the mean size of individuals (large animals are more vulnerable than small, or disperse) (Tainton & Mentis 1984). Fire reduces the diversity of invertebrate niches by reducing the strata to one, removing shelter from climate and predators, and eliminating detritus, resulting in the permanent disappearance of about 60% of the arthropod fauna from regularly burnt savanna. In South Africa, above ground invertebrates have many characteristics that allow them to escape from fire or survive in its wake: for example flying to safety, taking refuge or burying eggs in the soil, and high fecundity for population recovery from survivors in refugia (Tainton & Mentis 1984). In grasslands subject to greater seasonal influence than savanna, the effects of fire on invertebrates are compounded by the interaction of fire frequency and season of burning (Tainton & Mentis 1984).

South African grassland lizards and Australian mallee reptiles and amphibians survive fire by sheltering in holes, under rocks or in water (Tainton & Mentis 1984, Friend 1993). Birds escape by flying, and most ground-nesting species will re-nest, conditions permitting, if a clutch is destroyed by fire. South African and Australian small mammals suffer relatively high mortality, but their high fecundity allows relatively rapid population recovery. Increased vulnerability to predation because of reduced cover also has an impact on populations of small vertebrates which survive fire (Friend 1993). European mouse population irruptions usually follow fire in Australia, as do native mammal populations to a lesser extent, along with changes in species relative abundance (Friend 1993).

Wetlands

Many North American wetlands are burned periodically to improve habitat for wildlife, mainly for fur-bearing animals such as muskrat, and for geese and other game waterfowl (references in Kirby et al. 1988). Most studies indicate that reduction in height and density of tall emergent hydrophytes by fire and grazing benefits breeding waterfowl (Kantrud 1986). For example, fire in a wet prairie in Florida resulted in negligible bird mortality and some reptile mortality, but in the four months following, burned areas carried three times the bird population of adjacent unburned areas (Vogl 1973). However, burning to improve waterfowl habitat has been shown to reduce nesting habitat for herons and egrets (Bray 1984). Spring burns can destroy cover, kill ground nesting birds, and can reduce nest success (Kirkpatrick 1941). After early spring burning of waterfowl habitat in Iowa, USA, nesting success was low on burned plots and best in tall vegetation with greater cover (Messinger 1974). Biennial spring fire is used to manage marsh in southern USA for muskrats but if followed by overgrazing of burned areas by muskrats and geese, it creates habitat which favours waterfowl over muskrats (Allan 1950).

There is evidence that snails and periwinkles increase in density on southern and western USA coastal marshes following fire, but there is little other information on fire-invertebrate relationships (Komarek 1985). Matta & Clouse (1972) showed that the occurrence of adult forms of most insects apart from herbivorous katydids was not significantly reduced by fire in coastal wetlands in Virginia, USA.

New Zealand

In New Zealand there has been little research on the effects of fire on the native fauna, and most refers to invertebrates of tussock grasslands (e.g., Carlyle 1988).

6. Use of fire for management of natural areas

6.1 OVERSEAS EXPERIENCE OF PRESCRIBED FIRE FOR VEGETATION MANAGEMENT

6.1.1 *Development of prescribed fire policy and strategy*

Although fire has long been recognised as a natural phenomenon in rangelands and some forests, in the 1930s several economically catastrophic fires in the USA led to public demand for a policy of complete fire suppression. In 1935 the USDA Forest Service adopted a policy of containing fires within a day of ignition, and other agencies followed suit. By the 1970s this policy had become too expensive to maintain, especially in areas where the cost of fire suppression exceeded the value of the vegetation resource (Whitlock 1986). Now fire management policy takes into account this value, and calls for a balance between prescribed burning and limited suppression of wildfire (Butts 1983), the latter ranging from direct control to confinement, depending on management objectives (Towle 1983). In the USA federal system of Research Natural Areas, recognising the importance of fire as a major ecological process in many natural ecosystems has resulted in a change in policy from fire control to fire management (Johnson 1983). There remained, however, the philosophical dilemma of the purpose of the reserves: to preserve both natural conditions (structural maintenance) and natural processes (process maintenance). Conceptually, the latter has been adopted as the most appropriate management direction for Research Natural Areas. Prescribed fire is accepted as a means of meeting ecological objectives where reliance on natural fire is impractical, but the development of fire management policies and plans must be preceded by an evaluation of fire's role in a particular reserve (Johnson 1983).

The US National Park Service fire policy is to minimise as far as possible the influence of human-caused fires on the natural system. The suppression programme includes use of prescribed fires, both natural (fire distribution, extent, intensity and timing determined by natural systems of the particular Park) and resulting from a prescribed burn programme. The latter are primarily designed to restore natural fuel loadings and are only permitted in a dozen or so Parks under carefully constrained conditions and according to a specific burn plan (Butts 1983). They may also be used to reduce hazards along park boundaries and within developed areas of the park, and to manipulate park vegetation in limited areas to perpetuate species, vegetation types, or historical settings (Butts 1983).

In Uluru National Park, central Australia, deliberate fires were avoided for a century after European settlement, with the result that several wildfires burnt thousands of square kilometres of vegetation in the 1970s. A policy of deliberate burning has since been introduced in an attempt to return the

ecological condition that prevailed during aboriginal times (Griffin et al. 1986): a diverse but overall stable landscape in which fluctuations abound in a spatial and temporal mosaic.

6.1.2 *Applications of prescribed fire*

Prescribed burning has been defined as "the deliberate ignition of vegetation and the subsequent control of the limits of spread of the fire to achieve a desired management objective" (Hodgkinson et al. 1984) and "the scientific application of fire to wildland fuels under conditions of weather, fuel moisture, soil moisture, and other factors that allow the fire to be confined to a predetermined area, while at the same time accomplishing certain planned objectives" (Ford-Robertson 1971).

Australian conservation objectives of fire management (Green 1982, Parsons 1982, Good 1987) include:

- protection of user facilities and water catchments;
- protection of ecosystems where natural fire is rare;
- enhancing success of rare species by reducing competition from native species or exotic weeds;
- maintaining successional vegetation where this is only present within reserves;
- maintaining vegetation diversity to guard against catastrophic loss of particular vegetation types or species;
- wildfire hazard reduction;
- wildlife habitat improvement;
- preservation of naturally-functioning ecosystems;
- increasing water yield;
- elimination or maintenance of vegetation types or plant species.

All of these objectives are tempered by the fragmentation, distribution and extent of remaining natural areas.

In North America, prescribed burning of wetlands is used to halt succession to woody vegetation, to improve habitat for muskrats (which are economically important for their fur) and waterfowl (mainly for hunting), to control weeds, and to improve forage quality for cattle (references in Kirby et al. 1988). Surface burns are used to release early growing season plants for forage; root burns in dried substrate to control unproductive grasses and sedges; and peat burns, during drought, to convert marsh to open water (Uhler 1944).

6.1.3 *Planning prescribed fire regimes*

Knowledge of the natural fire regime of an ecosystem has been seen as a major issue in planning prescribed fire regimes. For example, Good (1987) stressed that a prescribed burning regime planned for a reserve should be based on research pertinent to conservation objectives, but that few data applicable to conservation reserve management were available on the effects of fires on vegetation and wildlife in Australian reserves.

In developing an expert system for fire management in Australia's Kakadu National Park, Haynes (1985) stressed the problems associated with selecting the "best" fire regimes for conservation purposes. Objectives could include achievement of "climax" vegetation, or maximum diversity, or optimising populations of rare or endangered species. Achievement of a pre-European fire regime would be constrained by inadequate information on the use of fire by pre-European people, imprecise knowledge of the pre-European vegetation, and inadequate control of the numerous ignition sources possible at present.

The ecological basis for management in Uluru National Park was determined from a map of vegetation units relating vegetation pattern to fire history. It was necessary to identify local vegetation pattern and spatial fuel characteristics that limit flame spread and to quantify fuel states and weather variables to model fire behaviour (Griffin et al. 1986).

A decision support system has been developed for generating fire management plans to achieve wildlife management objectives in an Australian conservation reserve (Baird et al. 1994). Plant species fire response curves to predict vegetation structure and wildlife habitat preference data provided the best available information upon which to frame decision rules.

6.1.4 Constraints on the use of prescribed fire

Depending on location and rainfall, negative effects of prescribed burning (Martin 1986) include:

- loss of forage;
- physiological damage to desirable plants;
- reduced vegetation and litter cover;
- accelerated runoff and erosion;
- post-fire overgrazing;
- invasion by weeds

In Australia the paucity of research information on fire management is a constraint to reliable advice to landholders about the timing and frequency of prescribed burning (Roberts 1986).

Invasion by fire-adapted weeds is a problem exacerbated by inappropriate fire management, but controlled fire can be used to reduce the impact of such weeds. For example, *Hakea sericea* in South African fynbos can be controlled by felling the plants, allowing release of seeds from the hard follicles, then burning 12 to 24 months afterwards to kill the new crop of plants before they reach reproductive age (Kruger 1981). However, this is not an effective control method because of the high cost, leading to unsustainable effort.

Gill (1977) cautioned that the long-term effects of chemical fire retardants with high nitrogen and phosphate content may have an impact on Australian native flora. Fire breaks create problems of soil stability, control of exotic species, and control of access, and may impact on the movement of fauna (Gill 1977).

6.2 USE OF PRESCRIBED FIRE FOR CONSERVATION MANAGEMENT OF NATURAL VEGETATION IN NEW ZEALAND

6.2.1 *History of prescribed fire*

Although deliberately lit but illegal fires are not uncommon, there is only a single record of the sanctioned deliberate use of fire in protected natural areas in New Zealand: an experimental spring burn of snow tussock grassland in the Flagstaff Scenic Reserve near Dunedin (Calder et al. 1992). However, this was undertaken to provide an experimental comparison with the effects of an autumn fire, rather than to manage vegetation for conservation purposes.

6.2.2 *Advocacy of prescribed fire*

Calder et al. (1992) used the results of the Flagstaff Scenic Reserve experiment to support their recommendation that snow tussock grassland at this altitude should be burnt every 15-40 years to maintain the community against scrub invasion.

Other authors also give consideration to the use of prescribed fire to maintain particular vegetation types in New Zealand protected natural areas. Wardle (1991) suggested that survival of gumland manuka heathland depends on acceptance of the use of fire for management. Rogers & Leathwick (1994) noted that deliberate burning may be condoned for sustainable management of red tussock grassland in Tongariro National Park, where it would otherwise disappear because of the spread of native shrubland and introduced heather. Miller (1993) suggested that fire could be used to maintain certain values of pakihi, and Williams et al. (1990) suggested that periodic burning of pakihi may be needed to ensure the availability of habitats for small herbaceous species by controlling the development of manuka stands.

6.2.3 *Constraints on the use of prescribed fire*

Miller (1993) was pessimistic that prescribed fire would achieve maintenance of the pakihi ecosystem in a desired static state, but did not elaborate on the reasons for this doubt. Williams et al. (1990) cautioned that a pakihi fire regime to conserve habitat for herbaceous plants would destroy fernbird (*Bowdleria punctata*) habitat. Echoing the reservations expressed by many overseas authors, Rogers & Leathwick (1994) pointed out costs and problems associated with deliberate burning of New Zealand vegetation, for example: difficulty in controlling burn patterns, vulnerability to weed invasion, ecosystem degradation, and social pressures for commercial exploitation. They stressed the ecological complexity of the effects of fire, and the importance of models of species recovery and invasion rates and patterns for determining fire frequency.

Weed invasion is the greatest threat to protected natural areas where vegetation is burnt. There is a substantial suite of naturalised species which have a range of fire adaptations. Many would prove ephemeral if the recovery of taller native vegetation could be assured, including herbaceous geophytic herbs such as

catsear and stoloniferous grasses such as browntop (e.g., Allen & Partridge 1988). However, a few herbaceous (e.g., mouse-eared hawkweed, Timmins 1992, Rogers 1994) and many woody (e.g., gorse and broom; Calder et al. 1992, himalayan honeysuckle (*Leycesteria formosa*); Timmins 1992, heather; Chapman & Bannister 1990) naturalised species can be tenacious occupiers of sites made available by fire. They can also completely displace the native vegetation of shrublands and grasslands, and profoundly affect the character of wetlands. Timmins & Mackenzie (1995) contains an exhaustive list of such weeds and their potential impact on biota and ecosystems, including their response to fire. A summary is given in Table 2.

6.3 FIRE BEHAVIOUR

Basher et al. (1990) summarised data on the character of grassland and shrub fires and how this influences burning impacts on plants and soils. Burning impacts vary according to the heat generated and intensity of fire, the vertical distribution of heat, the duration of heating, frequency of fire, and seasonality. In turn, these parameters are affected by the type, amount and disposition of fuel that has accumulated (species present, quantity of fuel, moisture and heat content of the fuel, the morphological and physiological adaptations of the plants present), weather conditions, moistness of soil, topography, wind, time of day and year, and the post-fire weather conditions and management. Fire is spatially and temporally variable in intensity and duration, resulting in patchy patterns of survival and recovery of vegetation, burning of litter, and heating of soil. Temperatures in grass and scrub fires typically range from 700 to 1000°C, although localised flame temperatures can be much hotter. Fire heat depends largely on the type of fuel available and can be greatly reduced when fuel is moist. In grassland fires, temperatures below the soil surface rarely exceed 100°C and temperature increases rarely occur below 10 mm. In scrub or shrublands, temperature rises can be more marked and persistent reaching 195°C at 25 mm below the soil surface and 50°C at 50 mm. Fires in grassland or grassland-scrub can be regarded as low-intensity with relatively low temperatures, short duration of temperature rise and rapid decrease of temperature below the soil surface.

Little information appears to be available on the character of fires in wetlands but it could be expected that they would be characterised by lower temperatures, shorter duration of elevated temperatures, and greater variation in heating, depending on quantity of fuel and variation in wetness within a wetland. For example, Clarkson and Stanway (1994) suggest fires at Whangamarino wetland were "cool" burns which destroyed live vegetation and litter but caused only slight heating of the underlying peat. A fire at Eweburn bog (Timmins, 1992) was characterised by impacts ranging from little vegetation or peat damage in areas with pools of open water to vegetation destruction and charring of peat in damp areas.

TABLE 2. FIRE-ADAPTED NATURALISED PLANT SPECIES IN PROTECTED NATURAL AREAS OF SHRUBLAND, GRASSLAND AND WETLAND, AND THEIR RESPONSE TO FIRE (FROM TIMMINS & MACKENZIE 1995).

SPECIES	VEGETATION	RESPONSE
<i>Agrostis capillaris</i>	grassland, shrubland, wetland	resprouts after cool fires
<i>Asparagus scandens</i>	shrubland	may resprout from tubers
<i>Berberis darwinii</i>	shrubland	resprouts from crown and roots
<i>Buddleja davidii</i>	shrubland	may resprout from crown
<i>Calluna vulgaris</i>	grassland, shrubland	resprouts, establishes seedlings
<i>Chrysanthemoides monilifera</i>	shrubland, gumland	seedbank; germination stimulated by fire
<i>Cortaderia jubata</i>	grassland, shrubland	resprouts, establishes seedlings
<i>Cortaderia selloana</i>	grassland, shrubland	resprouts, establishes seedlings
<i>Crataegus monogyna</i>	shrubland	resprouts after cool fire
<i>Cytisus scoparius</i>	grassland, shrubland	seedbank; germination stimulated by fire
<i>Erica lusitanica</i>	grassland, shrubland	resprouts, seedbank; germination stimulated by fire
<i>Hakea gibbosa</i>	shrubland, gumland	serotinous fruits; seed released by fire
<i>Hakea salicifolia</i>	shrubland	serotinous fruits; seed released by fire
<i>Hakea sericea</i>	shrubland	serotinous fruits; seed released by fire
<i>Hieracium pilosella</i>	grassland, some wetlands	low flammability, stoloniferous, responds to removal of competition
<i>Hieracium praealtum</i>	grassland, shrubland	low flammability, stoloniferous, responds to removal of competition
<i>Larix decidua</i>	grassland, shrubland	adults tolerate cool fire; establishes from seed on burnt areas
<i>Leycesteria formosa</i>	shrubland	resprouts from base after cool fire
<i>Ligustrum lucidum</i>	shrubland	may resprout from base
<i>Ligustrum sinense</i>	shrubland	may resprout from base
<i>Lupinus polyphyllus</i>	grassland, wetland	fire resistant foliage; may resprout; seedbank; establishment from seed
<i>Nasella trichotoma</i>	grassland	may resprout; germination from seed bank
<i>Pinus contorta</i>	grassland, shrubland	adults tolerate cool fire; serotinous cones release seed after fire
<i>Pinus nigra</i>	grassland, shrubland	adults tolerate cool fire; serotinous cones release seed after fire
<i>Pinus pinaster</i>	grassland, shrubland	adults tolerate cool fire; seeds invade burnt areas
<i>Pinus radiata</i>	grassland, shrubland	adults tolerate cool fire; seeds invade burnt areas
<i>Pseudotsuga menziesii</i>	grassland, shrubland	adults tolerate cool fire; seeds invade burnt areas
<i>Racosperma dealbatum</i>	shrubland	resprouts and spreads by suckers; seeds invade burnt areas
<i>Rhamnus alternifolius</i>	shrubland	resprouts
<i>Rosa rubiginosa</i>	grassland, shrubland	resprouts
<i>Rubus fruticosus</i>	shrubland	resprouts after cool fire
<i>Spartina alternifolia</i>	coastal wetland	resprouts from rhizomes
<i>Spartina anglica</i>	coastal wetland	resprouts from rhizomes
<i>Ulex europaeus</i>	grassland, shrubland, wetland	resprouts; seed dormancy broken by fire; huge seed bank of long-lived seeds
<i>Zizania latifolia</i>	wetland	resprouts from rhizomes

7. Issues and information requirements for use of fire in New Zealand protected natural areas

7.1 OBJECTIVES OF THE USE OF PRESCRIBED FIRE

Possible objectives for the use of fire as a conservation management tool on land administered by the New Zealand Department of Conservation are:

- maintenance of successional vegetation;
- maintenance of species diversity;
- maintenance of a particular plant community;
- maintenance of habitat for plant and animal species;
- reduction of fuel loadings to reduce fire hazard.

Before the use of prescribed fire is contemplated, answers to two questions are required:

- is prescribed fire desirable and feasible?
- where, when and how can prescribed fire achieve desired objectives?

Information from this paper should help answer the first question. The second requires much more detailed information and analysis, and will most likely be answered on a case by case basis. Further issues that will need to be addressed when considering fire in a particular reserve are:

- the purpose for which the reserve was established;
- clear and explicit definitions of fire management objectives;
- the role of natural fire in the reserve;
- any special resource and use considerations;
- the overall management objectives of the reserve and specific elements within it.

7.2 ASSESSMENT OF FIRE HAZARD

Not all vegetation types within a reserve contribute to hazardous fuel levels or ignition risks. Assessment of fire hazard and the need for hazard reduction burning require detailed knowledge of the fuel complexes of a reserve. In assessing the potential contribution of fuels to fire hazard all parameters which make up the fuel complex must be considered (e.g., litter production rate; proportions of litter, grass, and shrub foliage in total fine fuel; packing ratio; Good 1987). The fuel complex is therefore a function of the vegetation age and structure, the plant species present, the prevailing weather conditions, and the terrain. Classification and mapping by these factors are required to determine or predict fuel loads existing in an area to be burnt.

Fire hazard assessment also includes prediction of the potential for ignition and the regularity of fire occurrence at any site, as well as consideration of the geographical relationship of sites capable of carrying high intensity fire, the distance from sites or features of value, the potential rate of spread of a fire, and the suppression capability of the management agency.

7.3 PRESCRIBED FIRE PLANNING AND MANAGEMENT

Wardell-Johnson et al. (1989) noted that critical data requirements for fire management in reserves relate to four questions:

- what are the past, current and likely future fire regimes?
- how do the biota respond to these regimes?
- how does fire interact with reserve neighbours and visitors?
- what (fire management) activities can be undertaken with available resources?

Thus to develop and implement an effective fire management programme, it is necessary to identify management objectives, to understand the fire history and ecological role of fire in the area, and to know the effects of fire on the local biota (Parsons 1982). Information that will be needed in planning and decision making includes:

- locations of facilities and features that should not be burnt;
- location and nature of the fuel complexes of a reserve;
- proportion of area to be burnt and the proportion of fuel to be removed;
- maps and descriptions of the distribution and abundance of particular habitats that may be rich in fauna or contain key faunal species;
- inventories of threatened animal and plant species whose populations could be reduced or eliminated by burning;
- identification of weeds which may be adapted to fire or are potential occupiers of sites made available by fire;
- environmental impacts of various fire control methods e.g., fire suppressants and retardants or clearing of fire breaks.

The determination of whether a fire poses an acceptable or unacceptable risk is subjective. Fire risks continually change in space and time in response to weather, fuel, public/neighbour expectation, and use. In any one fire not all areas of the fire pose the same threat, even though the hazards (fuel load etc.) may be similar. Managers must identify the level of risk a facility or feature of value can be exposed to, and the limits of risk they are prepared to accept (Kessell and Good 1985). The risk assessment components of fire management programmes must also recognise the economic and biological value of the resources and features which may be affected by fire (Good 1987).

Case studies of fire management planning processes are available from overseas, particularly Australian, sources. Kilgore (1983), Lotan et al. (1983) and Burrows et al. (1987) contain several papers which address the issue of fire management in reserves from philosophical, policy and applied viewpoints. Although these issues are not further considered in the present review, we recommend that these publications be consulted in the course of policy development for fire management in New Zealand reserves.

8. Monitoring fire impacts and recovery

8.1 MONITORING APPROACHES

Monitoring is required to assess and evaluate the success of fire management practices in achieving their objectives. It involves the identification of undesirable effects and research needs, and leads to the refinement of burning strategies (Wouters 1992). Specific management needs are usually readily identified during the planning of a fire programme and can become the focus of monitoring, but monitoring should also provide long-term information to allow improvement of fire management.

There are two main approaches to monitoring fire impacts. The first is chronological series comparisons (inferring the nature of fire impacts from sites that have been burnt at different times in the past), which have the advantage of rapidly providing a broad overview of long-term trends, and tend to be less expensive since they can be carried out over a short time period. However, this approach assumes that the plant communities were similar before the fire, and that the fire characteristics and post-fire conditions of each fire were similar. It tends to miss the short-term changes to soils and fauna that occur within the first year of burning, and there are difficulties in selecting comparable sites as well as comparable plant communities. It has been used in studies of fire impacts on snow tussock communities New Zealand (e.g., Gitay et al. 1992).

The second is single site monitoring through time, which requires vegetation and site characteristics to be determined prior to burning, and the nature of fire impacts is directly monitored at intervals following a fire. With this approach, there is a clear cause-and-effect relationship between observed changes and burning, and it is possible to document both short-term and long-term changes. However, it takes much longer to yield meaningful information on long-term trends, and tends to be more expensive as monitoring sites need to be maintained over long periods.

A combination of both approaches will be most useful to determine both the short and long-term impacts of burning.

Soil monitoring should be governed by the following principles:

- most changes will be in the top few mm to cm of soil, so sampling needs to be concentrated at shallow depths (perhaps sample litter, 0-2, 2-5 and 5-10 cm depths)
- many of the effects (particularly biological) will be most marked in the 1-2 years after burning, so monitoring should be most intensive immediately after burning, with subsequent longer intervals between measurements;
- potential changes are:
 - physical (water repellency, structural decline/crusting, soil temperatures, cover/litter, sheet and wind erosion),

- chemical (nutrient budgets, especially for C, N and S but also P and cations, and including vegetation, litter and soil),
- biological (micro-organisms and associated nutrient pools, microbial biomass and activity, invertebrates - soil and litter).

Important factors to consider for monitoring the biota are changes in abundance of both dominant and fire-sensitive species, changes in species composition, invasion by weeds, changes in the area and diversity of each plant community, vegetation age distribution and, in relation to the survival of fauna, quantity and location of fire refugia and potential threats from predators and competitors (Wouters 1992 and references therein). To obtain strategic management information about particular species, Wouters (1992) recommended monitoring of the means of regeneration (seed or sprout), the time between fire and production of an adequate seed supply, and the time between fire and the disappearance of the species from the site.

Hopkins (1987) stressed the importance of establishing a monitoring programme to provide reliable information upon which to base a fire management regime. He noted that the field component of monitoring cannot function in isolation, but must be used in conjunction with a range of activities from data management to a review of results, policies and management programmes. He suggested that the first essential function of monitoring is archival, to maintain adequate records of decisions and actions. Data processing should include a prompt for sampling time falling due. He commented that photographic records have very serious limitations for collection of quantitative data.

8.2 MONITORING TECHNIQUES

The use of monitoring to record changes in New Zealand vegetation is well established (e.g., references in Allen 1993a, Allen 1994). However, there are doubts that some methods provide useful data, and the results of many monitoring studies are unreported (Allen 1993b). Allen (1994) provided guidelines for the selection and use of monitoring techniques, most of which are applicable for recording changes following fire in shrublands, grasslands and wetlands. These are summarised in Table 3.

TABLE 3. ADVANTAGES AND LIMITATIONS OF MONITORING METHODS, AND THEIR APPLICABILITY TO MONITORING ON THE CONSERVATION ESTATE (FROM ALLEN 1994).

METHOD	ADVANTAGES	LIMITATIONS	APPLICATION ON DOC ESTATE
intercept	<ul style="list-style-type: none"> -simple to do with little equipment - widely used - measures cover 	<ul style="list-style-type: none"> -site choice requires experience -requires plant identification skills -line intercept more time consuming and less accurate than point intercept, and rarely used 	<ul style="list-style-type: none"> -all objectives except population trends of threatened plants -helps understand cause/effect of change -basis for management decisions
plot/transect	<ul style="list-style-type: none"> -simple to do with little equipment - widely used - measures frequency and density -estimates cover 	<ul style="list-style-type: none"> -site choice requires experience -requires plant identification skills 	<ul style="list-style-type: none"> -all objectives -helps understand cause/effect of change -basis for management decisions
height frequency	<ul style="list-style-type: none"> -becoming widely used -gives records of both composition (species frequency) and structure (biomass index) of vegetation 	<ul style="list-style-type: none"> -requires special equipment -site choice requires experience -can be slow and tedious to do -requires plant identification skills 	<ul style="list-style-type: none"> -all objectives -helps understand cause/effect of change -basis for management decisions
quadrats	<ul style="list-style-type: none"> -simple to do with little equipment - widely used - measures abundance/dominance, frequency and density 	<ul style="list-style-type: none"> •site choice requires experience -requires plant identification skills -abundance/dominance recording not common in NZ 	<ul style="list-style-type: none"> -all objectives -helps understand cause/effect of change -basis for management decisions
photographs	<ul style="list-style-type: none"> - moderately simple to do -reduced field time 	<ul style="list-style-type: none"> -requires special equipment and some photographic skills -plant identification difficult -accurate measurements difficult - measurement removed from field situation 	<ul style="list-style-type: none"> -demonstration purposes such as evidence or displays
marked plants	<ul style="list-style-type: none"> -simple to do with little equipment 	<ul style="list-style-type: none"> -limited application of results 	<ul style="list-style-type: none"> - monitoring of population trends only -helps understand cause/effect of change -basis for management decisions

9. Recommendations

9.1 FIRE MANAGEMENT POLICY

Fire management policy should be based on:

- an understanding of the fire history and fire ecology of the area/community/species for which fire management is proposed;
- an understanding of the short- and long-term effects of fire on the biota and soils of the area;
- a clear definition of the objectives of fire management;
- an understanding of the methods by which the objectives can be achieved.

9.2 PRIORITIES FOR FIRE CONTROL

Priorities for fire control should be based on the possible outcomes of fire. Social considerations such as the protection of life, property, or landscape values are outside the scope of this report. For the purposes of biological conservation, under the following circumstances, in descending order of priority, fire should be contained and extinguished:

- the purpose of the protected natural area is the conservation of a particular organism or group of organisms (e.g., the Cromwell chafer beetle; Ward & Munro 1989);
- plant, vertebrate or invertebrate species listed in the New Zealand Threatened and Local Plants List (Cameron et al. 1995) or the Department of Conservation's conservation priority list (Tisdall 1994), and not known to be fire tolerant, are present;
- fire-adapted naturalised species potentially taller than the natural vegetation are present or could invade (Timmins & Mackenzie 1995);
- plant and animal communities are present which are rare in the protected natural areas of the ecological region and which are not fire-adapted;
- the area contains vegetation which, in the absence of fire, will be succeeded by vegetation which is under-represented in the protected natural areas of the ecological region (e.g., kanuka with regenerating podocarps);
- the area contains mosaics or sequences of vegetation which reflect environmental variations such as altitudinal or edaphic gradients which are rarely found together in the protected natural areas of the ecological region.

9.3 USE OF FIRE

Priorities for fire use should be based on the possible outcomes of failure to burn. Again, only biological conservation criteria are considered here: landscape and other social values should be the subject of a separate discussion. The

circumstances under which fire could be used to manage vegetation of protected natural areas for biological conservation, in descending order of desirability, are as follows:

- to enhance success of rare species by reducing competition from native species or exotic weeds, and thus to ensure the conservation of plant or animal species which would otherwise be rare in or absent from the ecological region or New Zealand (e.g., orchids in cushion bog; Clarkson & Stanway 1994);
- to maintain successional vegetation where this might otherwise become rare in or absent from the protected natural areas of the ecological region (e.g., pakihi; Miller 1993);
- to maintain species diversity in fire-prone vegetation within a particular protected natural area so that re-establishment of a natural succession remains possible after accidental fire;
- to maintain vegetation diversity to guard against catastrophic loss of particular vegetation types or species from the ecological region;
- to maintain successional vegetation for scientific or educational purposes;
- to reduce fuel load in fire-prone vegetation to avoid the possibility of an intense and destructive accidental fire.

9.4 FURTHER RESEARCH

Little or nothing is known of the response of most New Zealand ecosystems and individual plant and animal species to fire. With the probability of climate change and the certainty of increasing public use of protected natural areas, the risk of fire is continually increasing. Successful conservation of fire-prone natural areas depends on an understanding of the effects of fire. We recommend the establishment of a research programme to fulfil the following objectives:

- to determine the adaptations and response to fire of individual species of fire-prone ecosystems;
- to determine plant community responses to fire at different seasons and frequencies, and of different intensities, in fire-prone ecosystems;
- to determine the response of fire-adapted naturalised species to fire at different seasons and frequencies, and of different intensities, in fire-prone ecosystems;
- to determine if prescribed fire could be used to reduce fuel loadings without compromising species diversity and ecosystem processes in fire-prone ecosystems.

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