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Fire management implications of fuel loads and vegetation structure in jarrah forest restoration on bauxite mines in Western Australia

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Abstract

Bauxite mines being restored to native jarrah (*Eucalyptus marginata*)-dominated forest in Western Australia have accumulated substantial fuel loads. Assessment of fire management aspects is necessary for the effective integration of restored areas into forest-wide management. Fuel characteristics, vegetation structure and fire behaviour of young (5- and 8-year-old) restored bauxite mines in Western Australia were examined. Pre-burn fuel loads were moderate in 5-year-old restoration (15.0 t ha^{-1}) and high in 8-year-old restoration (29.8 t ha^{-1}). Large ranges in available fuel load estimates of sample sites (2.2–60.8 t ha^{-1}) indicated the heterogeneous nature of fuel distribution leading to variable fire behaviour. The vegetation structure of restored areas differed from that of the unmined jarrah forest due to the presence of a prominent mid-storey layer composed of number of acacias (*Acacia pulchella*, *A. celastrifolia*, *A. extensa*, *A. drummondii* and *A. lateriticola*). This mid-storey layer contributed 49% of the total fuel load in 5-year-old restoration, although the high proportion of live material (73.6%) inhibited fire development. In 8-year-old restoration the mid-storey layer contributed 46% of the total fuel load. The lower proportion of live material in these sites (27.9%), due to the senescence and death of the relatively short-lived acacias, led to increased fire intensities, flame heights and higher levels of crown scorch and defoliation. Prescribed burns were conducted in early summer 1997. Burns in 5-year-old restoration were of low intensity (<250 kW m^{-1}) while burns in 8-year-old restoration were of very high intensity (>7000 kW m^{-1}). Fuel re-accumulation was rapid in the first 2 weeks post-burn, with litter-fall rates 2–3.5 times that of unburnt control sites. Thereafter, litter-fall and fuel accumulation in burnt restored and unmined sites was comparable to that of unburnt control sites. Analysis of 1 and 2 years post-burn 5-year-old restoration indicated that the prescribed burns had failed to remove the mid-storey acacia layer and actually increased the proportion of dead standing material, whereas in the 8-year-old restoration, the prescribed burns removed the mid-storey layer of acacia shrubs and stimulated an increase in the proportion of live plant material, particularly near ground level. Maximum soil temperatures recorded by heat sensitive crayons exceeded 300 °C in 8-year-old restoration burns but were less than 100 °C in 5-year-old restoration burns. Hard-seeded species were stimulated to germinate in the top 2–3 cm of the soil following burns in 8-year-old restoration and only in the top 0.5 cm of the soil following burns in 5-year-old restoration. Some seeds in the top 1 cm of the soil may also have been killed by the high temperatures generated in burns in the 8-year-old sites. Fire management of jarrah forest

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restoration will have to be different to that employed in unmined jarrah forest due to differences in fuel characteristics, vegetation structure and fire behaviour.

Keywords: *Acacia*; *Eucalyptus*; Fire management; Fuel loads; Jarrah forest; Litter-fall; Mining restoration; Vegetation structure

1. Introduction

The fire regime of an area describes the frequency of fire occurrence, the season of occurrence and its intensity (Gill, 1981). All three factors vary although they are generally interlinked. Fire frequency is largely determined by potential ignition sources, the prevailing weather conditions of an area and the time taken to build up levels of available fuel after fire (Whelan, 1997). The frequency at which fires occur has the potential to alter the floristic composition of vegetation communities (Morrison et al., 1995; Bradstock et al., 1997). Species that regenerate by seed after fire are generally the most sensitive to fire frequency, as they require sufficient time between fires to reach reproductive maturity and to replenish their seed banks (Bell et al., 1989; Bond and van Wilgen, 1996; Whelan, 1997). If fires occur too frequently then seeder species may become locally extinct (Fox and Fox, 1986). Seeder species are favoured in habitats with long intervals between fires, while species that resprout following fire tend to be favoured in habitats with frequent fires (Bell et al., 1989; Bell, 2001).

Seasonal burning effects have been reported in many Mediterranean systems including the fynbos of South Africa, Californian chaparral, French garrigue and Australian dry sclerophyll forest (Bond and van Wilgen, 1996; Whelan, 1997). In southern Australia, fires in late summer or autumn are typically more intense than fires in spring (Burrows, 1987, 1994; Whelan and Tait, 1995; Auld and Bradstock, 1996). This is generally due to lower fuel moisture contents after the long summer drought. High intensity fires can result in greater post-fire seedling establishment due to the greater removal of inhibiting trash and litter layers (Facelli and Kerrigan, 1996; Grant et al., 1997a; Whelan, 1997). In Mediterranean climates, autumn burns are quickly followed by the rainy season that provides seedlings with 5–6 months of moist, mild conditions to establish deep tap-roots before the onset of the next summer. In contrast, spring burns are followed by 4–6 months of hot, dry conditions. Over this period, dormant seeds are vulnerable to insect predation and soil pathogens (Majer, 1980; Smith et al., 2000).

Fire intensity is an indicator of energy output and can be classified as low (<500 kW m^{-1}), moderate (501–3000 kW m^{-1}), high (3001–7000 kW m^{-1}) and very high (>7000 kW m^{-1}, Cheney, 1981). Fire intensity is usually determined by the quantity of fuel available, its moisture level and the rate at which it combusts (Sneeuwjagt and Peet, 1985). Higher available fuel levels and lower fuel moisture levels generally result in increased fire intensity (Cheney, 1981; Sneeuwjagt and Peet, 1985; Burrows, 1994).

Vegetation structure is known to affect fire behaviour in the forests of southwest Western Australia (Sneeuwjagt, 1971; Sneeuwjagt and Peet, 1985; McCaw, 1986). The vertical and horizontal distribution of fuel can, in some instances, be more important than available fuel loadings in determining fire behaviour (Gould, 1993; Gould et al., 1997). The vertical distribution of fuel in sclerophyll forests is important as crown fires generally only develop where there is a continuous fuel profile (Chandler et al., 1983). Where there is only low understorey vegetation it takes a fire of greater intensity to develop into a crown fire.

Fuel loads and vegetation structure are the only site components that can be directly manipulated to influence fire regime and this is the rationale behind much of the prescribed burning that is conducted in Australian forests (Bell et al., 1989). The jarrah (*Eucalyptus marginata*) forest of Western Australia is managed by the Department of Conservation and Land Management (CALM). Since 1954, CALM has used widespread low intensity controlled burning as a key management tool to reduce fire hazard in the jarrah forest (McCaw and Burrows, 1989). The principle objective of fuel reduction burning is to limit the potential intensity of unplanned fires and to provide greater scope for fire suppression (McCaw et al., 1992).

This is achieved by reducing the quantity of fuel and by modifying its spatial distribution. The interval between successive prescribed burns in Western Australia's multiple land-use forests varies according to the rate of fuel accumulation, silvicultural objectives and proximity to property and other assets (McCaw et al., 1996). Currently, the jarrah forest is burnt in a mosaic pattern on a rotation of 5–12 years (Burrows, 1985). In 1999–2000, a total of 174,455 ha of forest and associated vegetation types was prescribed burnt in southwest Western Australia. The majority of planned burning was conducted in spring (69%) and autumn (23%, CALM, 2000).

Alcoa World Alumina Australia (Alcoa) has conducted bauxite mining and restoration in Western Australia's jarrah forest since 1963. The current objective of the restoration is to establish a self-sustaining jarrah forest ecosystem, planned to enhance or maintain water, timber, recreation and conservation values. Approximately 500 ha of forest are cleared, mined and restored each year at Alcoa's two operating mines. Areas restored by Alcoa in the jarrah forest now exceed 11,000 ha. Early restoration techniques (1966–1976) involved planting exotic pine trees or non-indigenous eucalypts with no understorey species. From 1976 to 1987, restored sites were planted with a mix of native and eastern Australian eucalypt species and seeded with a dense native species understorey. Since 1988, a native overstorey has been established in restoration sites by seeding jarrah, marri (*Corymbia calophylla*) and blackbutt (*Eucalyptus patens*). Alcoa's current restoration practices are successful in re-establishing up to 100% of the species present in the pre-mining forest (J.M. Koch, unpublished). Floristic similarity of older restored sites (pre-1988) to the native forest are lower (Nichols and Michaelsen, 1986) due to the intentional establishment of non-native overstorey species, longer stockpiling of topsoil and the application of a less diverse understorey seed mix (Grant and Loneragan, 1999).

In 1997, Alcoa and CALM developed agreed completion criteria and standards for the restoration of bauxite-mined land in the jarrah forest. Five broad principles were established that restored areas would need to meet to be considered successful and hence complete. These are that restored areas: meet land use objectives; be integrated into the natural landscape; exhibit sustained growth and development; have vegetation as resilient as the jarrah forest; and be capable of integration into wide-spread forest management practices (Elliott et al., 1996). Specifically, the completion criteria require restored areas to be resilient to fire and capable of integration into CALM's jarrah forest fire management program. Although Alcoa manages and improves its restoration areas through the implementation of an active research and development program (Nichols et al., 1985; Nichols, 1998) the company has historically maintained a fire exclusion program. This was due to a lack of knowledge on the long and short-term effects of fire on restoration areas (J.M. Koch, pers. comm.). A number of recent studies, however, have investigated various aspects of fire ecology and fuel load management on pre-1988 restoration sites (Collins, 1996; Grant et al., 1997a, 1998). The aim of this study was to document aspects of fuel load, vegetation structure and fire behaviour in 5- and 8-year-old restoration to provide a basis for effective fire management of current restoration protocols.

2. Methods

2.1. Site description

The study sites were located at Alcoa's Jarrahdale Bauxite Mine (32°17′S, 116°08′E, Fig. 1), approximately 45 km SSE of Perth, Western Australia. Jarrahdale mine was closed in 1998 and final restoration was completed in 2001. The climate is typically Mediterranean with winter rainfall and summer drought. Mean annual rainfall at the site is 1187 mm. Mean maximum temperatures of 30 °C are experienced in the hottest months of January and February. Mean minimum temperatures range from 15.7 °C in February to 6.1 °C in August. The soils are pisolithic or massive laterite over a layer of kaolinitic clay (Koch, 1987). The vegetation of the study area consists of dry sclerophyll open forest dominated by the overstorey species jarrah and marri. There is a middle-storey consisting of bull banksia (*Banksia grandis*), sheoak (*Allocasuarina fraseriana*) and snottygobble (*Persoonia longifolia*) (Dell et al., 1989). Economically significant deposits of bauxite ore are present in the soil profile of ridge tops and valley slopes.

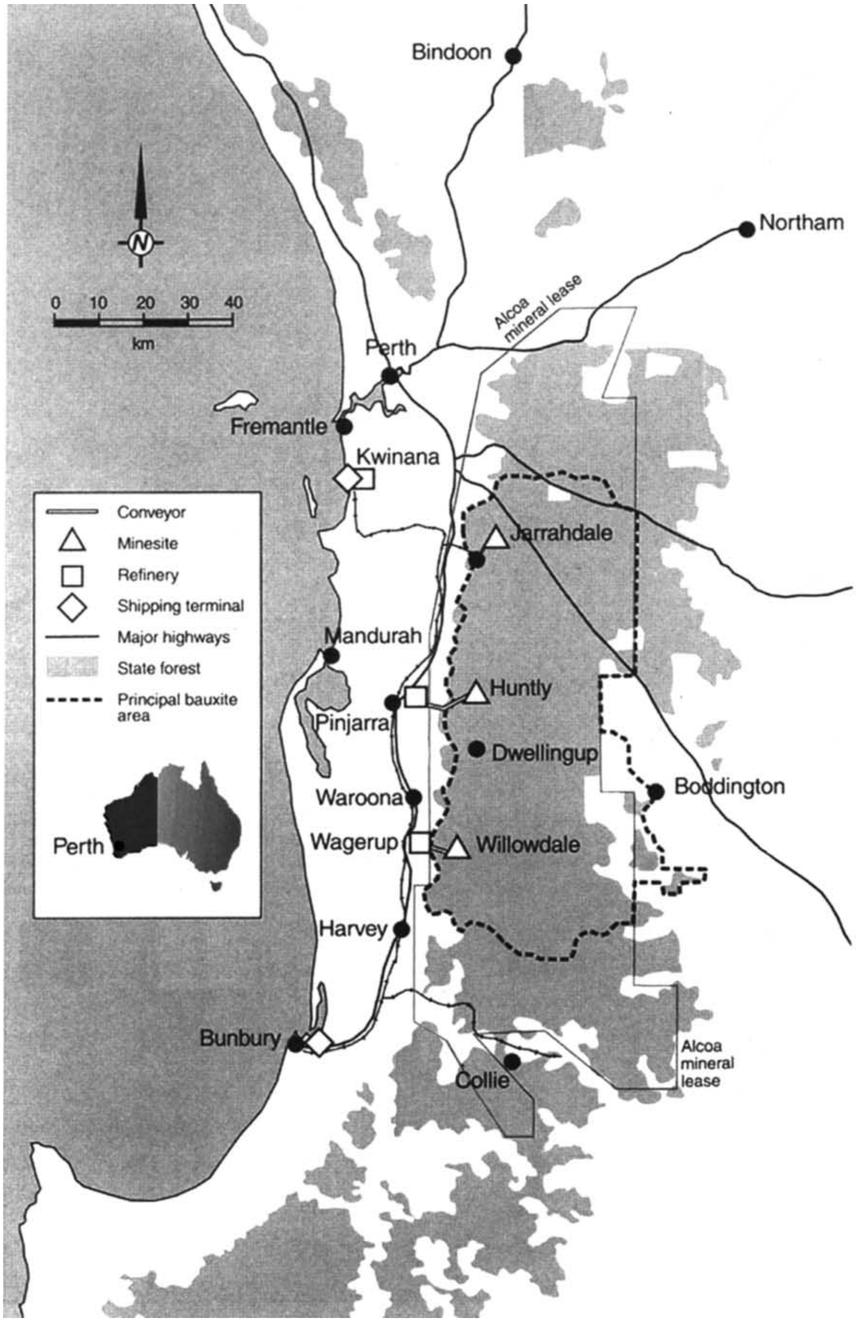

Fig. 1. Location of the Jarrahdale study site in southwest Western Australia. Adapted from Ward et al. (1996).

2.2. Mining and restoration

Bauxite extraction in the jarrah forest is conducted as shallow strip mining. Mining occurs in isolated pods of 1–100 ha in area (av. 10–20 ha) where the alumina grade is economically viable (>27.5%). Prior to mining, sawlogs, minor forest produce (e.g. fence posts) and firewood are removed and the remaining vegetation cleared. The gravelly sand covering the bauxite is then stripped and used for restoration. Where possible the topsoil, which contains much of the soil nutrients, organic matter, seeds and micro-organisms, is used immediately after stripping to restore a nearby pit (direct return). Where logistics or forest disease considerations dictate, the topsoil is stockpiled.

Following removal of the bauxite, the overburden and topsoil are replaced and ripped to a depth of 1.5 m to alleviate soil compaction. A seed mix containing 60–80 native understorey and eucalypt species is applied in autumn. A fertiliser mix of di-ammonium phosphate with added potassium and micronutrients (16% P, 14% N, 5% K plus Cu, Zn, Mn and Mo) is applied by helicopter at the rate of 500 kg ha^{-1} in spring (Ward et al., 1996).

2.3. Study sites

Fieldwork was conducted between August 1997 and February 2000. Six mining pits restored in 1992 (5-year-old) and six restored in 1989 (8-year-old) were selected at Alcoa's Jarrahdale mine site in 1997. Six permanent transects were also established in adjacent unmined native jarrah forest of variable fire history (9–18 years since last burn). These forest sites provided a good comparison to the restored areas because jarrah forest sites of variable fire history that are found in the same area tend to be floristically more similar to each other than geographically separated sites with the same fire history (Bell and Koch, 1980). In all three regions, two burn treatments were investigated: an early summer burn and no burn (control). Each burn treatment was monitored using a permanent transect (128 m) in each of three replicate pits. The dominant overstorey species in the 5- and 8-year-old restoration were jarrah, marri and blackbutt. The understorey vegetation cover was dominated by live and senescent acacias but also contained representatives of many other plant families especially those of the Asteraceae, Myrtaceae, Papilionaceae and Rhamnaceae.

2.4. Fuel load and composition

Prior to burning, available fuel loads were estimated by collecting plant material to a height of 3.9 m, from five quadrats (1 m × 0.5 m) randomly positioned along each of the permanent transects. In restored sites, quadrats were placed at right angles to the rip-lines to ensure both a ridge and a furrow were sampled. Litter was collected separately from standing material. Standing material was sorted into two categories: trash (dead material <25 mm in diameter, McCaw, 1986); and live (living material <4 mm in diameter, Burrows, 1994). The maximum diameter (25 mm) of material considered as trash fuel is higher than Burrows (1994) recommended for native jarrah forest. This increased diameter was used because studies indicated that more of the standing fuel is likely to be consumed in burns conducted in restored areas because it is well aerated and often dead (Grant et al., 1997a). Following collection, fuel samples were oven dried at 80 °C for 48 h to obtain an estimate of available fuel load. Available fuel loads were also estimated at one month and 1 and 2 years after the burns.

Ground litter cover was measured by estimating the percentage cover of litter in 128 contiguous quadrats (1 m × 1 m) along each transect. Estimates of litter cover were conducted before burning and 1 and 2 years after burning.

2.5. Vegetation structure

The horizontal and vertical distribution of standing fuel at each site was characterised using a modified levy pole method (Sneeuwjagt and Peet, 1985; Collins, 1996; Grant et al., 1997a). The levy pole was 3.9 m in height (1 cm diameter) and marked into 30 cm sections. The pole is placed vertically and the number of vegetation contacts within each 30 cm height section recorded. Contacts were classified as live or dead, and whether they were a tree or understorey species. Along each permanent transect 320 levy pole placements were made prior to burning, for a total of 960 levy pole placements in each burn treatment of each vegetation type. These measurements were repeated 1 and 2 years after the burns.

2.6. Litter-fall

Litter deposition was estimated by collecting debris in traps placed in permanent locations. Litter-fall traps consisted of plastic trays (34 cm × 28 cm, area = 0.0952 m^2) with fine drainage holes in the base.

Ten litter-fall traps were placed along each transect, making a total of 30 traps in each burn treatment of each vegetation type, and 180 traps over the entire experiment. Traps were deployed, in both burnt and unburnt control sites, 1 day after completion of the prescribed burns. Debris material that fell into the traps was collected at the end of every month, with the first collection on 31 December 1997 (2 weeks post-burn). Monthly collection continued for 2 years. Collected material was sorted into four categories: leaves, twigs and bark, reproductive material (e.g. buds, flowers and fruits) and trash (e.g. dead insects and unrecognisable material <5 mm in diameter). Material was oven dried at 80 °C for 48 h to obtain dry weights of litter-fall. Weights were divided by trap areas to estimate litter-fall rates.

2.7. Fire behaviour

Officers from the Department of Conservation and Land Management (CALM) recorded the soil dryness index (SDI), the rate of spread (ROS) of the main fire front and flame height during the prescribed burns. Weather information (temperature, humidity, wind speed) was obtained from meteorological stations at Jarrahdale and Karnet (10 km south of Jarrahdale, Bureau of Meteorology, 1997). Byram's (1959) fire intensity (I, kW m^{-1}) was calculated using a simplification of the standard equation (Burrows, 1994):

$$I = (W - r)\,\mathrm{ROS}\,0.516$$

where W is the total fuel <25 mm in diameter (t ha^{-1}), r the fuel residue (t ha^{-1}) and ROS the rate of spread (m h^{-1}).

2.8. Soil temperatures

Heat-sensitive crayons with melting temperatures of 45, 66, 121, 204 and 302 °C were drawn on 10 cm × 10 cm pieces of Hardifence™ fibre board that were placed vertically in the soil (known as thermocolour pyrometers, Hobbs et al., 1984). Six pyrometers were placed in each site prior to burning. Following the burns, the boards were removed and the depth of heat penetration into the soil was estimated from the melting of the crayons.

Soil temperatures during burns in the 8-year-old restoration were also recorded by four automatic data loggers (Unidata Australia Portable Starlogger™). Data loggers were positioned adjacent to where the samples used to estimate fuel load were collected. Data loggers were placed in pairs with recordings taken from adjacent rip-line ridges and furrows, as fuel loads often differ between these two locations. Four thermistors (range: 10–100 °C) were attached to each data logger and buried at depths of 1, 2.5, 5 and 10 cm. Mean temperatures were recorded every 10 s with maximum temperatures logged every 60 s.

3. Results

3.1. Pre-burn fuel characteristics and vegetation structure

Average pre-burn available fuel loads increased with age in jarrah forest restoration (Fig. 2). Fuel loads were moderate (15.0 t ha^{-1}) in 5-year-old restoration and high (29.8 t ha^{-1}) in 8-year-old restoration. Fuel loads were heterogeneous, within and between pits, ranging from 2.2 to 38.7 t ha^{-1} in 5-year-old restoration and from 12.6 to 60.8 t ha^{-1} in 8-year-old restoration. Fuel loads in unmined jarrah forest, unburnt for 9–18 years, were high (21.8 t ha^{-1}) and also variable (10.0–36.8 t ha^{-1}). Litter was the major source of fuel in all three vegetation types, comprising 51.1 and 54.4% in 5- and 8-year-old restored sites, respectively, and 68.8% in the unmined jarrah forest. Standing fuel in 5-year-old restoration was dominated by live material (73.6%), while in 8-year-old restoration only 27.9% of standing material was live.

Ground litter coverage was lowest in the pre-burn 5-year-old restoration at 31.6% (Table 1). By 8 years, the cover was almost double the 5-year level. In comparison the unmined jarrah forest averaged 88.3% litter cover.

Vertical fuel distribution in restored areas differed markedly from that of the unmined jarrah forest (Fig. 3). Restored areas generally had a uniform fuel distribution while unmined forest sites had a two-tiered vegetation structure, with high fuel density below 0.6 m and low fuel density above. Restored areas had a prominent mid-storey layer composed of live and senescent acacias that was lacking in the unmined jarrah forest. The prominent mid-storey

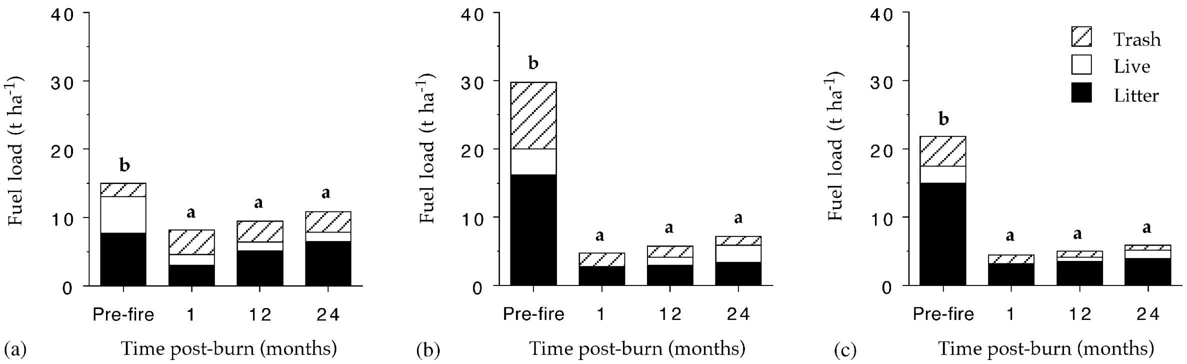

Fig. 2. Total available fuel load in (a) 5-year-old restoration, (b) 8-year-old restoration and (c) unmined jarrah forest. Fuel loads were recorded prior to burning and 1 month, 1 and 2 years post-burn. Different letters represent significant difference via Fisher's protected least significant difference (95%).

contributed to a near continuous fuel layer extending from the ground to the top of the tree canopies in restored areas. In the unmined forest there was a substantial separation of the mostly ground-based fuel from the tree canopy (10–20 m above). There were also differences in vegetation structure between the restoration sites of different ages. The fuel structure of 5-year-old restoration was dominated by live material, with only the 0–0.3 m level having more dead than live material. Above 2 m there was almost no dead fuel in 5-year-old sites with ratios of live to dead material exceeding 20:1. The fuel structure of 8-year-old restoration had more dead than live material below 1.5 m, indicating the development of substantial trash fuel loads. Five-year-old restoration had high proportions of shrub material below 1 m in height, however, above this height the fuel was dominated by tree foliage. In contrast, 8-year-old restoration was dominated by shrub material to a height of 3 m, with tree foliage predominating above. This is indicative of the lower tree canopy height and lower biomass of acacias in 5-year-old compared to 8-year-old restoration.

Table 1
Percentage of ground covered by litter (fuel connectivity) in restored and unmined jarrah forest sites, before and after burning[a]

Site type	Litter ground coverage (%)		
	Pre-burn	1 year post-burn	2 years post-burn
5-Year-old restoration	31.6 ± 3.8 a	45.2 ± 4.0 b	57.5 ± 8.1 c
8-Year-old restoration	63.7 ± 1.5 b	12.3 ± 2.8 a	13.7 ± 1.3 a
Jarrah forest	88.3 ± 3.2 a	nr	70.7 ± 9.3 a

[a] Values are mean ± S.E. (nr: not recorded). Within rows different letters represent significant difference via Fisher's protected least significant difference (95%).

3.2. Fire behaviour

Three burns were conducted in this study, one in each of the three vegetation regions examined. All burns in the current study were conducted over an 18 h period and under a similar SDI. Burns in 5-year-old restoration were of low intensity (Cheney, 1981), while those in 8-year-old restoration were of very high intensity (Table 2). The unmined jarrah forest sites were burnt by low to moderate intensity fires. The effects of the different components of wind speed, temperature at lighting, and humidity on fire intensity were not examined in detail due to the small number of burns undertaken. Prescribed burns in 5-year-old restoration and the unmined jarrah forest were conducted within the guidelines recommended for burning in the jarrah forest (Sneeuwjagt and Peet, 1985). These generally involve low wind speeds at the time of lighting, a falling hazard (i.e. burning in the evening when temperatures are falling and humidity is rising) and achieving rates of spread conducive to even fire intensities. Similar lighting techniques were used, with lines of fire being run parallel to each other. In the jarrah forest, fire-lines were lit at spacings of 10–30 m. These fire-lines met at a time of

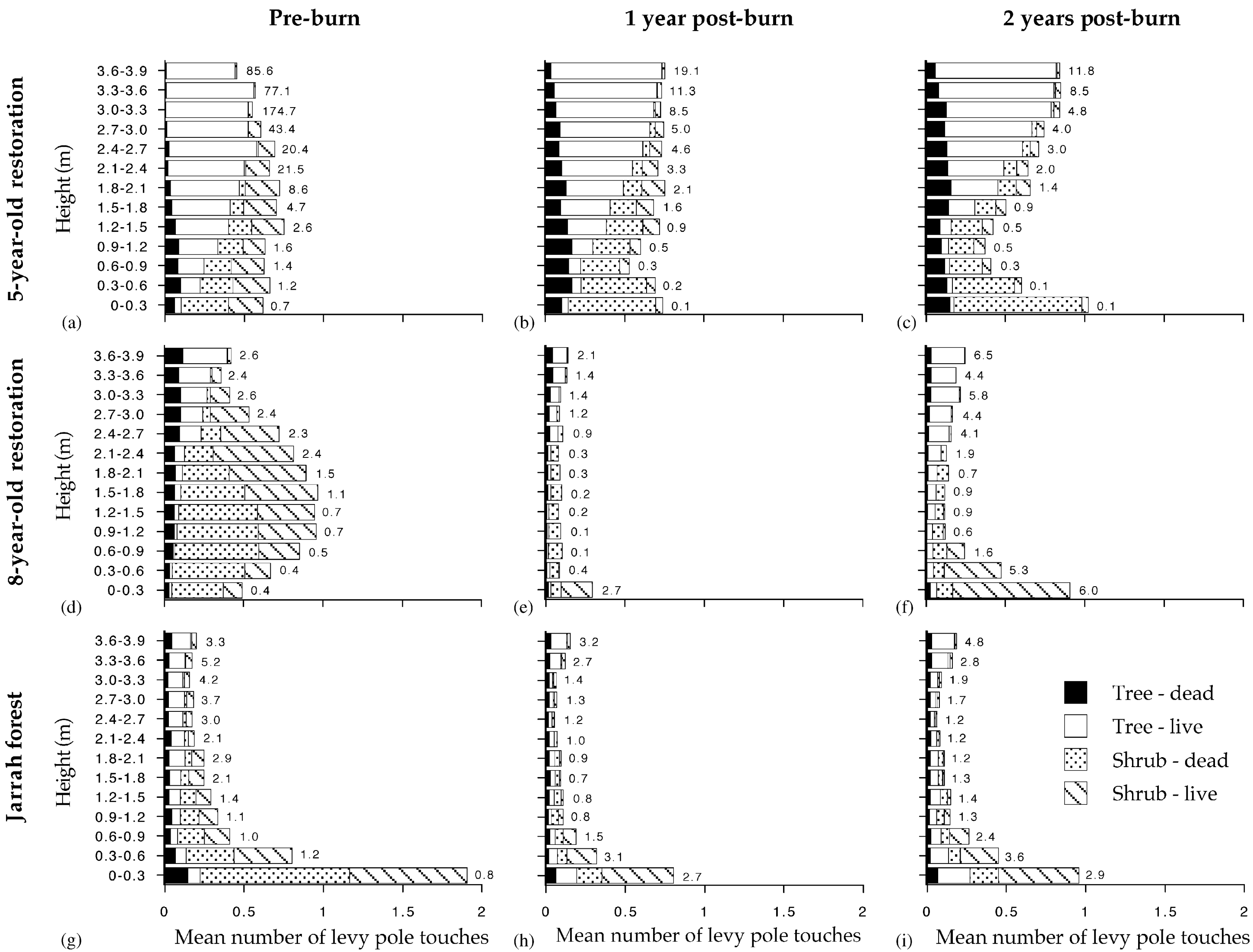

Fig. 3. Fuel structure estimated using the levy pole method for (a–c) 5-year-old restoration, (d–f) 8-year-old restoration and (g–i) unmined jarrah forest. Data are for pre-burn and 1 and 2 years post-burn. Solid bars represent dead tree material, open bars represent live tree material, open hatched bars represent dead shrub material and shaded bars represent live shrub material. Numbers on bars are proportions of live to dead plant material in each 0.3 m height category.

Table 2
Summary of fire information collected for the prescribed burns conducted in 5-year-old (1992) and 8-year-old (1989) restoration and unmined jarrah forest[a]

Site type	1992	1989	Jarrah forest
Date of burn	13 December 1997	12 December 1997	12 December 1997
Age at burn (years)	5	8	9–18
Time of ignition	10 a.m.	3 p.m.	3–7 p.m.
Wind speed (km h^{-1})	5–10	0–5	0–5
Temperature at lighting (°C)	23	31	31–26
Relative humidity at lighting (%)	70	41	41–60
SDI	1595	1584	1584
ROS (m h^{-1})	0–60	200–1200	15–200
Fire intensity (kW m^{-1})	0–232	2673–16049	130–1786
Flame height (m)	0.5–1	2–20	0.5–4
Canopy scorch (%)	3	43	2–10
Canopy defoliation (%)	0	57	0

[a] The SDI is a scale of moisture levels and ranges from 0 (field capacity) to 2000 (maximum soil dryness).

falling hazard and flame heights were mostly low (<1 m).

In 5-year-old restoration fire-lines were initially lit 10 m apart, however, due to the lack of continuous litter and poor fire spread, the litter in each rip-line was individually lit. Despite this, the fire only 'trickled' along the fuel in the rip-line furrows. As a result, 5-year-old restoration sites burnt very patchily, with only about half of the actual area burning. Flame heights were generally low (<1 m) and tree canopy scorch was minimal (Table 2).

Burns in 8-year-old restoration were ignited using a line of fire through the centre and then a ring of fire around the perimeter. The centre fire drew air in from the edges of the pits, thus drawing in the perimeter fire. This lighting technique was used to obtain an indication of the response of restored areas to very high intensity fires. Ten minutes after ignition, wind speeds rose from 0 to 5 km h^{-1} to an estimated 20–30 km h^{-1}. This wind was generated as air was drawn in from the burn perimeter to replace air lost through convection above the fire. The increased wind speed lowered the flame angle and caused 'spotting' of fires ahead of the flame front. The 8-year-old restoration experienced extreme fire behaviour with a crown fire and flame heights reaching 15–20 m. These sites experienced high levels of canopy defoliation (57%) and all trees not defoliated, mostly around the pit edges, were heavily scorched (Table 2).

3.3. Soil temperatures

Maximum soil surface temperatures measured during fire by thermocolour pyrometers were all below 121 °C in 5-year-old restoration, but exceeded 302 °C in 8-year-old restoration (Fig. 4a). Deeper heat penetration through the soil profile was recorded following burns in 8-year-old compared to 5-year-old restoration. Temperatures recorded for fires in the unmined jarrah forest were intermediary between those of the restored sites, with maximum soil surface temperatures exceeding 121 °C and moderate heat penetration into the soil profile.

Soil temperatures recorded by data loggers in 8-year-old restoration showed differences between adjacent ridges and furrows (Fig. 5). Soil temperatures on ridges exceeded 100 °C for 6 min at a depth of 1 cm, while the maximum temperature at 2.5 cm was 49.9 °C. In furrows, the maximum temperature at a soil depth of 1 cm was 80.4 °C, however, temperatures reached 57.4 °C at a depth of 2.5 cm. The mean duration of heating above 60 °C at 1 cm was 22 and 24 min in ridge and furrow sites, respectively (Fig. 5). At soil depths of 2.5 cm, temperatures did not exceed 60 °C, a result that supports measurements of the thermocolour pyrometers. The higher soil temperatures prior to burning on ridges compared to furrows are due to the ridges being exposed to the sun, while litter shades the furrows.

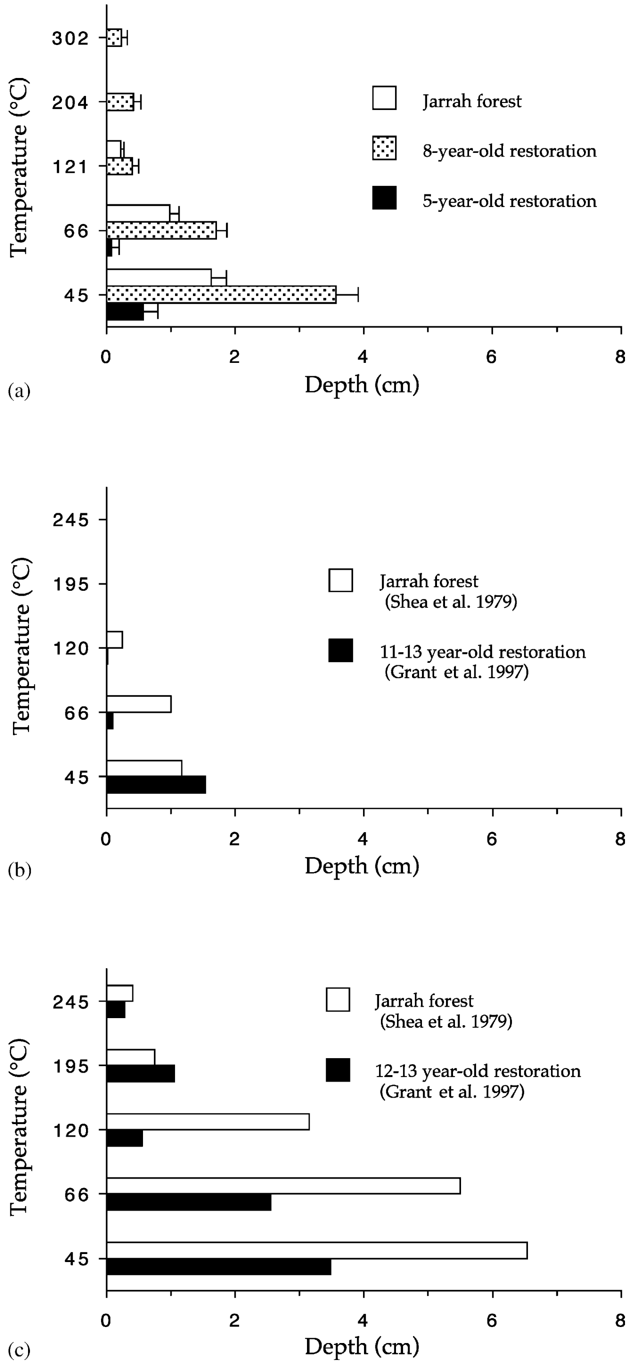

Fig. 4. Soil temperature–depth profile measured using thermocolour pyrometers for (a) early summer burns conducted in this study, (b) spring burns and (c) autumn burns conducted in 11–15-year-old restoration (Grant et al., 1997a) and jarrah forest (Shea et al., 1979). Bars are means ± S.E.

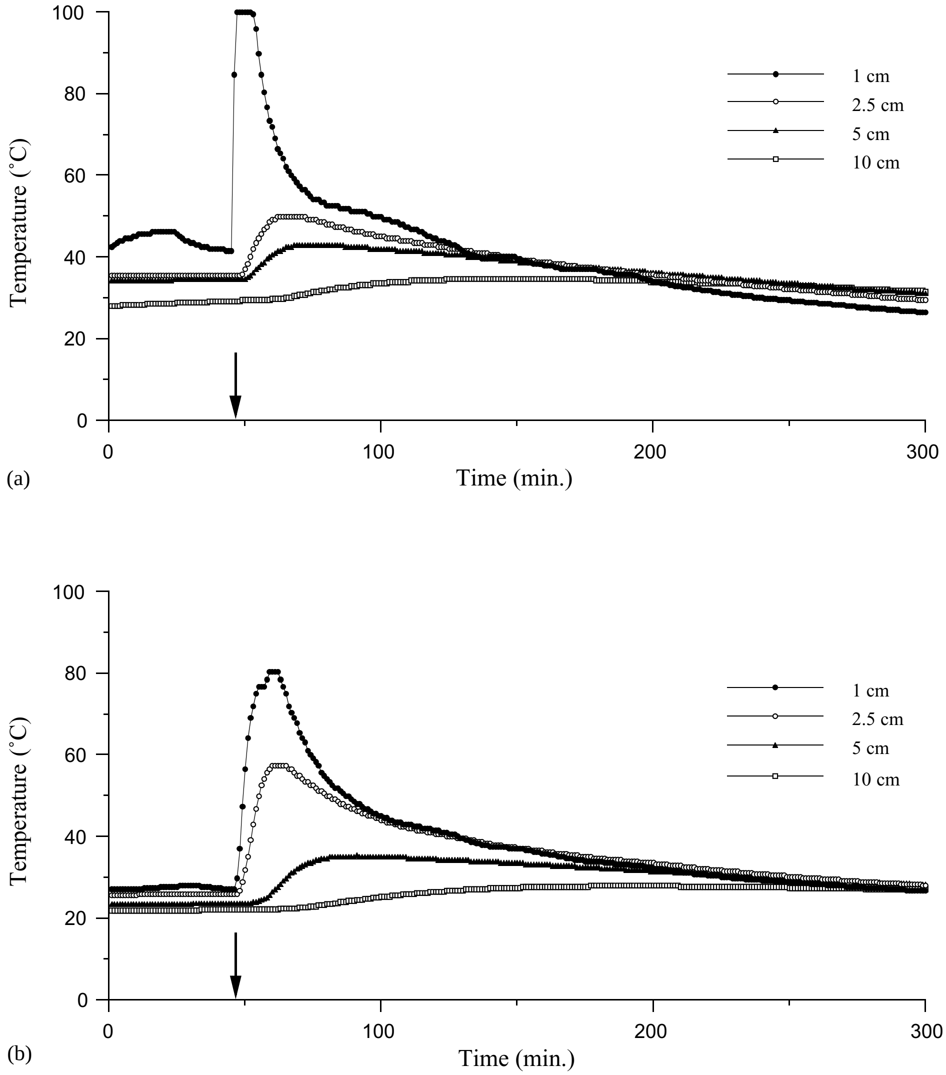

Fig. 5. Temperature traces recorded by data loggers in 8-year-old restoration for a rip-line (a) ridge and (b) furrow. The arrows indicate arrival of the fire front above the thermistors.

3.4. Post-burn fuel characteristics and vegetation structure

Prescribed burning significantly reduced fuel loads in all sites (Fig. 2). One month post-burn, available fuel loads had been reduced by 46, 84 and 79%, relative to pre-burn levels, in 5- and 8-year-old restoration and unmined jarrah forest, respectively. Fuel load heterogeneity was also greatly reduced after burning in 5-year-old restoration (pre-burn $s^2 = 331$, post-burn $s^2 = 85$), in 8-year-old restoration (pre-burn $s^2 = 1111$, post-burn $s^2 = 38$) and in unmined jarrah forest (pre-burn $s^2 = 575$, post-burn $s^2 = 42$). Fuel load re-accumulation was rapid within the first month post-burn but was generally slower thereafter. Five-year-old restoration had higher post-burn fuel loads than 8-year-old restoration and considerably greater ground litter cover (Fig. 2, Table 1). This was mainly

Table 3
Mean annual litter-fall deposition in restored and unmined jarrah forest sites[a]

Site type season	5-Year-old restoration ($t\ ha^{-1}$ per year)	8-Year-old restoration ($t\ ha^{-1}$ per year)	Jarrah forest ($t\ ha^{-1}$ per year)
Summer	1.832 ± 0.114 a (50.9)	1.661 ± 0.067 a (48.0)	1.811 ± 0.078 a (53.1)
Autumn	0.661 ± 0.061 b (18.4)	0.712 ± 0.167 b (20.6)	0.586 ± 0.050 b (17.2)
Winter	0.290 ± 0.027 c (8.1)	0.322 ± 0.039 c (9.3)	0.323 ± 0.043 c (9.5)
Spring	0.814 ± 0.054 b (22.6)	0.763 ± 0.059 b (22.1)	0.689 ± 0.071 b (20.2)
Total	3.597 ± 0.159 (100)	3.458 ± 0.145 (100)	3.409 ± 0.128 (100)

[a] Values are mean litter-fall ($t\ ha^{-1}$ per year) ± S.E. for unburnt (control) sites for each season over a period of 2 years. Seasons are: summer (December–February), autumn (March–May), winter (June–August) and spring (September–November). Values in brackets are percentages. Within columns different letters represent significant difference via Fisher's protected least significant difference (95%).

due to less complete litter combustion and increased post-burn leaf-fall in 5-year-old restoration, while much of the canopy in 8-year-old restoration had been completely defoliated by the burn (Table 2).

Mean annual litter-fall in unburnt control sites was similar for both restored and unmined jarrah forest sites (3.409–3.597 $t\ ha^{-1}$, Table 3). Litter-fall in all sites was dominated by leaf material, comprising 65 and 59% in 5- and 8-year-old restoration and 53% in the unmined jarrah forest (Fig. 6). The proportion of reproductive material in the litter-fall of restored sites remained relatively constant with age (14–17%), however, bark and twigs comprised a greater proportion in 8-year-old compared to 5-year-old restoration. Litter-fall was strongly seasonal in both restored and unmined jarrah forest sites (Table 3), being highest in summer (December–February) and lowest in winter (June–August).

Following burning, litter remained the major source of fuel in all three site types (Fig. 2). Immediately after burning there was a large pulse of litter-fall in all sites. In 5- and 8-year-old restoration, respectively, litter-fall in the 2 weeks period following burning was 3.5 times (0.586–2.078 $t\ ha^{-1}$) and 2 times (0.539–1.093 $t\ ha^{-1}$) that of the control sites. The post-burn litter-fall pulse in the unmined jarrah forest was also high (3.4 times the control, 0.630–2.148 $t\ ha^{-1}$). Following this immediate post-burn input, litter-fall deposition in burnt sites remained comparable to that of the unburnt control sites. Burning altered the composition

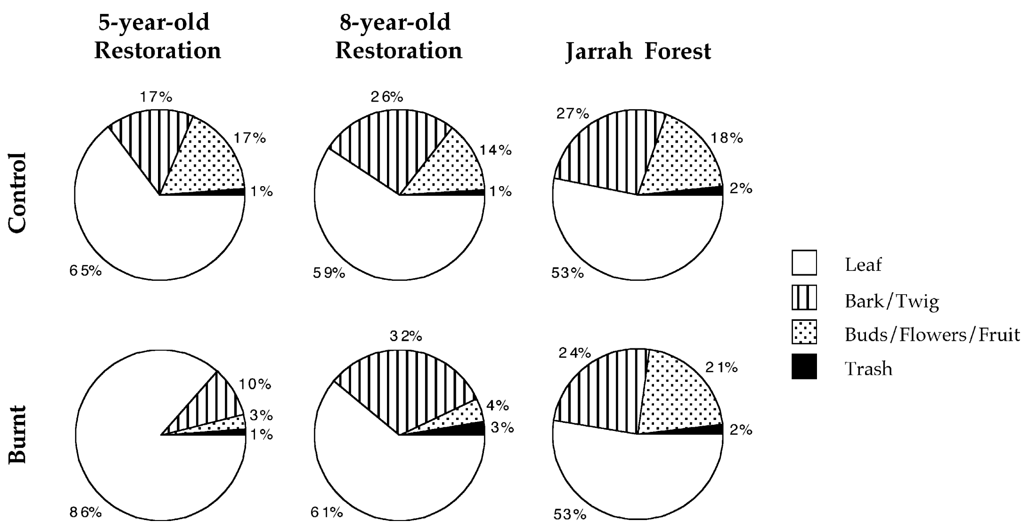

Fig. 6. Litter-fall composition in restored and unmined jarrah forest sites. Graphs are means of 2 years of data from either burnt or control sites.

of litter-fall in restored sites, but induced little change in litter-fall composition of the unmined jarrah forest (Fig. 6). The proportion of leaves increased in 5-year-old restoration, bark and twig material increased in 8-year-old restoration, while reproductive material decreased in both the 5- and 8-year-old restoration.

One year post-burn, standing fuel in 5-year-old restoration showed a large increase in the proportion of trash material (Fig. 2). The vegetation structure of

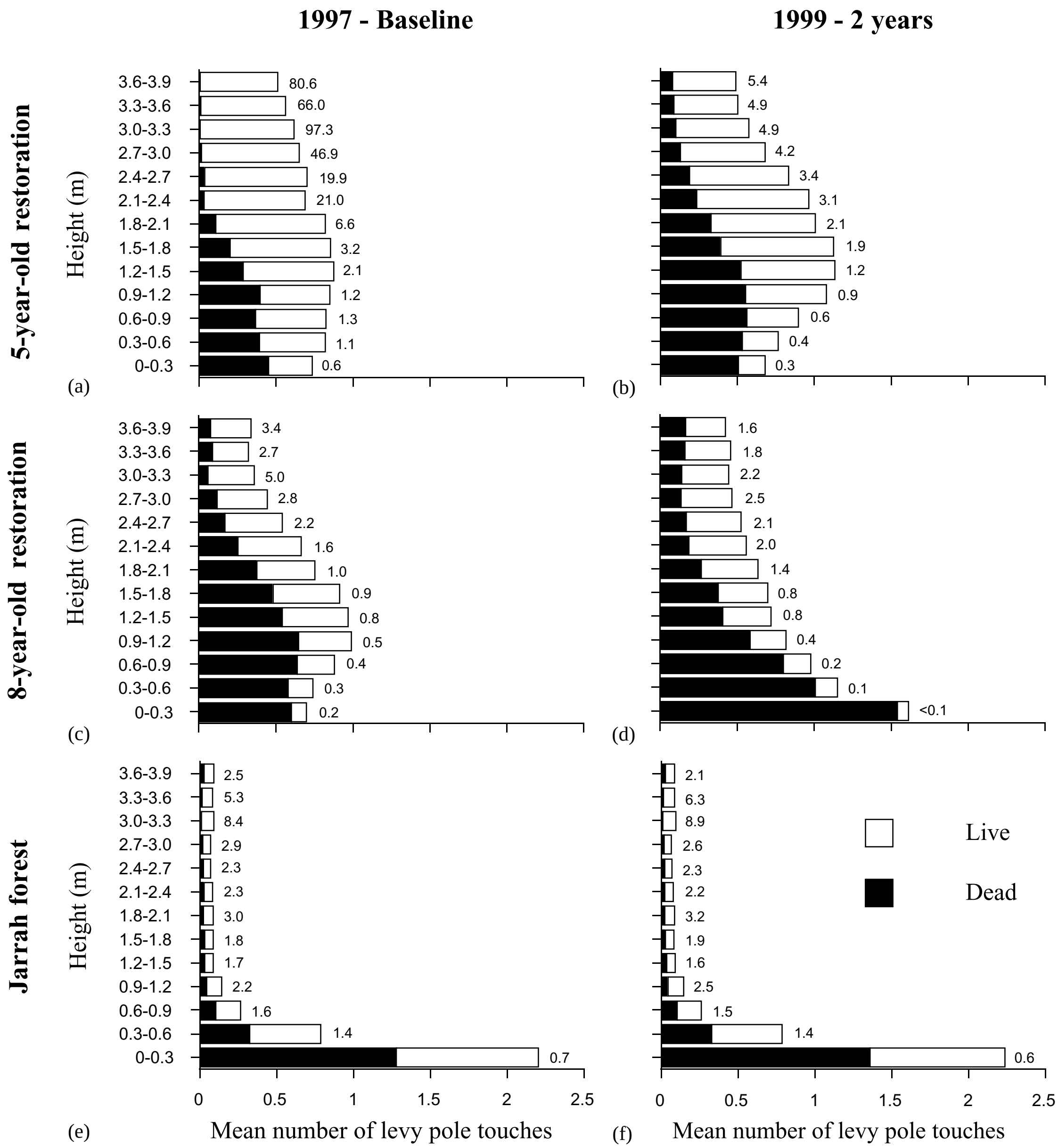

Fig. 7. Fuel structure of control sites measured using the levy pole method for (a and b) 5-year-old restoration, (c and d) 8-year-old restoration and (e and f) unmined jarrah forest. Data are for baseline and 2 years later. Solid bars represent dead plant material and open bars represent live plant material. Numbers on bars are proportions of live to dead plant material in each 0.3 m height category.

the 5-year-old restoration at this time differed little from its pre-burn vegetation structure (Fig. 3). However, by 2 years, a greater proportion of the shrub component was dead and tree foliage was less abundant lower in the vegetation structure. In contrast, all standing fuel in the 8-year-old restoration was dead (trash) after 1 month post-burn. From 1 month to 2 years after burning in 8-year-old restoration the proportion of live standing material increased rapidly due to post-fire seedling establishment and plant regeneration via resprouting. Vertical fuel distribution in these older restoration sites, 1 and 2 years after burning, differed markedly from the pre-burn fuel distribution. One year post-burn, the quantity of both dead and live plant material was substantially reduced. By 2 years post-burn there was an increase in live material, particularly below 0.6 m, due to shrub regeneration, and above 2 m, due to tree resprouting. The vegetation structure of 8-year-old restoration and unmined jarrah forest was similar 2 years after burning.

In the absence of fire, vegetation structure in 5-year-old restoration becomes similar to that of pre-burn 8-year-old restoration (Fig. 7). In turn, 8-year-old restoration began to show fuel accumulation near ground level as the trash layer collapsed.

4. Discussion

4.1. Pre-burn fuel characteristics and vegetation structure

Available fuel loads in young (5- and 8-year-old) jarrah forest restoration increased with age, a result consistent with previous studies of mine revegetation development in Australia (Collins, 1996; Ward and Koch, 1996; Grant et al., 1997a; Chaffey and Grant, 2000) and will continue to increase until equilibrium is established between fuel deposition and decomposition rates (McCaw et al., 1996). Mean fuel loads of 8-year-old restoration in the current study (29.8 t ha^{-1}) are comparable to that of 11–15-year-old (pre-1988 Alcoa restoration) restored jarrah forest incorporating non-indigenous eucalypt tree species (28.5 t ha^{-1}, Grant et al., 1997a). In comparison, the quantity of fuel on the jarrah forest floor in unmined areas reaches an equilibrium of 16–18 t ha^{-1} within approximately 25 years after burning (Bell et al., 1989; Burrows, 1994). Fuel loads in restored eucalypt forests were still increasing 20 years after completion of restoration procedures (Collins, 1996). If this trend is continued in the current jarrah forest restoration it is possible that, in the absence of management intervention, mean fuel loads could exceed 60 t ha^{-1} within 20 years. This situation would present a considerable fire hazard, and reinforces the need to develop fire management protocols for restored ecosystems on mined lands.

Spatial distribution of fuel within restored areas was heterogeneous and produced diverse fire behaviour within any one burn. Fuel loads can vary substantially over very small distances as a result of differential accumulation of fuel in the ridges and furrows of rip-lines. Such large variation in fuel loads was often evident within distances of less than 1–2 m. The senescence and death of acacias can create high, localised fuel loads. In contrast, rip-line ridges are often bare or very sparsely covered by litter. Spatial heterogeneity of fuel loads has the potential to influence fire behaviour in restored areas (Grant et al., 1997a). Patchy fuel levels may allow vegetation to remain unburnt or burn at a lower intensities than elsewhere. This may be advantageous as it could provide refuges for animals or fire-sensitive plants and may aid re-colonisation after burning (Whelan, 1997). Conversely, however, heterogeneous fuel distribution may allow for patches of high intensity fire in a burn of overall low to moderate intensity. This may provide sufficient soil heating, in places, to stimulate germination of hard-seeded species, while being of low enough intensity for fire-sensitive species to survive. Thus heterogeneous fuel loads may provide benefits for diverse post-fire plant regeneration and establishment. However, heterogeneous fuel loads can produce unpredictable fire behaviour that may increase the difficulty of attaining burning objectives and require greater management effort and planning to ensure safe outcomes.

The composition of fuel in jarrah forest restoration changed significantly with age. Although the proportion of fuel as ground litter remained fairly constant there were important changes in the composition of standing fuel. Standing fuel in jarrah forest restoration primarily consists of large acacias and low-level tree foliage. As restored areas mature, the tree canopy grows taller and shrubs become the major component of fuel. After approximately 6–15 years, tree canopies in restored areas have shed their lower leaves and acacias start to senesce and die. This produces dense thickets of dry, well-aerated elevated fuel. In addition, litter-fall from the tree canopy often results in large amounts of dry leaves and twigs being suspended atop of this standing material. Comparisons between 5-year-old and 8-year-old restoration showed that the younger sites had proportionately less dead material (26%) than the older sites (72%), because few acacias had senesced by 5 years of age.

The fuel structure of jarrah forest restoration contrasts markedly with that of the unmined jarrah forest. Restored sites had a more uniform vertical and horizontal fuel distribution, reflecting the greater density of mid-storey shrubs. The jarrah forest, in contrast, lacked a mid-storey shrub layer and exhibited a pronounced concentration of plant material near ground level. Ground level fuel in the jarrah forest tends to be continuous, mainly as litter (Burrows and McCaw, 1990; Burrows, 1994). In restored areas, where ground fuel may be discontinuous (due to rip-lines), the elevated trash layer can sustain fire by providing a continuous layer of aerated fuel. Due to the lower moisture content of the trash layer it can be consumed at lower combustion temperatures than would be required to burn the litter layer. Combustion of elevated fuel commonly results in increased fire intensities and scorch heights (Raison et al., 1983) and is often responsible for assisting fire into the tree canopy (Burrows, 1997; Grant et al., 1998).

4.2. Fire behaviour

This study has shown that fire can provide an effective means of reducing fuel loads in 5–8-year-old jarrah forest restoration. The burns were undertaken in dry fuels (15.0–29.8 t ha^{-1}), two to three times the maximum level recommended for prescription burning in jarrah forest (8 t ha^{-1}, Burrows, 1994). In dry sclerophyll forests of eastern Australia available fuel loads of 5–12 t ha^{-1} are presumed to be the threshold level above which fire could become intense enough to move into the tree canopy (Adams and Simmons, 1996; Vines, 1983). It is therefore likely that fuel loads in the current study are beyond the level that would result in controllable fire behaviour under moderate fire weather conditions. All of the prescription burns, however, were completed safely without the fires becoming uncontrollable. This is significant as the ability of fire personnel to safely operate in the jarrah forest is paramount. Burning of restored areas will reduce fuel loads, which decreases wildfire risk in these areas and may have positive effects on the vegetation (Grant et al., 1997b).

The SDI is a model that estimates the dryness of the soil profile and litter material of the jarrah forest; SDI and fire intensity are significantly correlated (Hobbs and Atkins, 1990; Burrows, 1994). All burns in the current study were conducted under a similar SDI. Fire intensity, however, was much higher in 8-year-old restoration than 5-year-old sites. Grant et al. (1997a) also found that in restored jarrah forest areas SDI was not correlated to fire intensity. The lack of correlation between SDI and fire intensity in restored vegetation was primarily related to fuel structure and connectivity.

The equation used to measure fire intensity includes a variable for the ROS of fire. Slower moving fires are of lower intensity for the same amount of fuel consumed. In 5-year-old restoration, the factor that most affected ROS was fuel discontinuity. Even though these sites contained more than 7 t ha^{-1} of litter on the ground, fire only ‘trickled’ along the rip-line furrows. The lack of fuel connectivity (32% ground fuel cover) did not allow a single fire front to burn and, as a result, the burn could not gain in speed or intensity. In contrast, 8-year-old restoration with its prominent trash layer and greater fuel connectivity (64%) was able to attain much greater ROS. Winds generated by burns in 8-year-old restoration would have also increased the ROS by increasing oxygen supply to the fire, contributing to greater mixing of combustion gases, increasing pre-heating of the fuel bed by lowering of the flame angle and by spotting ahead of the main fire front (Burrows, 1994).

4.3. Soil temperatures

In the jarrah forest, soil heating during burning is related to the amount of fuel burnt, and the moisture content of the soil and the fuel (Christensen and Kimber, 1975; McCaw, 1988; Burrows, 1994). In general, soil heating is reduced where fuel levels are low or where the soil and litter are moist. In 11–13-year-old jarrah forest restoration significant differences in surface soil heating depended on the season of burn (Grant et al., 1997a). Spring burns (when fuels were moist) resulted in lower soil temperatures and reduced soil heat penetration compared to sites burnt in autumn (drier fuels). Burns in the current study were conducted in early summer, when fuels were dry.

A comparison of soil temperatures obtained in the current study burns with those obtained in 11–13-year-old restoration fires (Grant et al., 1997a) and unmined jarrah forest prescription burns (Shea et al., 1979) indicates that soil temperatures in the 5-year-old restoration and unmined jarrah forest were similar to spring burns (Fig. 5). Soil temperatures in the 8-year-old restoration were similar to autumn burns. Maximum soil surface temperatures were below 100 °C in 5-year-old restoration but exceeded 300 °C in 8-year-old restoration. Reduced soil heating in 5-year-old restoration was probably due to lower fire intensity, shorter residence time and fuel bed continuity.

In jarrah forest sites the greater fuel depth meant that soil was still moist underneath. Under such conditions only the dry surface fuel burns resulting in reduced heat penetration of the soil (Burrows, 1994). Surface temperatures in jarrah forest fires can be very high (300–500 °C), while shallow depths (15 mm) remain considerably cooler (av. 40 °C, Burrows, 1994). Furthermore, while soil surface temperatures exceeded 300 °C in a mild intensity jarrah forest burn, temperatures at soil depths of 1.5 and 3 cm only reached 45 and 40 °C, respectively (Koch and Bell, 1980). Burn temperatures in 8-year-old restoration and 12–13-year-old restoration (Grant et al., 1997a) sites were lower than those obtained for a jarrah forest autumn burn (Shea et al., 1979). This may be due to the absence of large fuel components (e.g. dead fallen logs) in both restored areas. Dry logs can remain ignited for extended periods of time, resulting in localised areas of very high soil temperatures and heat penetration relative to the rest of the burn (Gill, 1981).

Soil heating by fire has significant implications for subsequent post-fire plant establishment. About 30% of jarrah forest species depend on seed stored either in soil or in the canopy for regeneration following death of the parent plant (Christensen and Kimber, 1975; Bell and Koch, 1980; Bell et al., 1993). The topsoil seed bank of the restoration contained 25–30 seeds m^{-2} of hard-seeded species (mostly legumes and *Trymalium ledifolium*, Smith et al., 2000). The majority of these seeds occur in the top 5 cm of the soil profile where they are stimulated to germinate (Shea et al., 1979; Portlock et al., 1990; Bell et al., 1993; Bell and Williams, 1998). Ambient temperatures of over 55–60 °C result in the cracking and increased germination of *Acacia pulchella* seed (Portlock et al., 1990), similar results were found for 35 leguminous species in eastern Australia (Auld and O'Connell, 1991). Moist-soil, spring burns in jarrah forest would have to burn in excess of 14 t ha^{-1} of fuel before sufficient soil heating occurred to stimulate germination of *A. pulchella* at a depth of 10 mm under standard fuel and soil types (Burrows, 1999). Dry summer or autumn burns in contrast would require the combustion of only 5 t ha^{-1} of fuel.

Five-year-old jarrah forest restoration burns showed very little heat penetration of the soil and, therefore, it is unlikely to be adequate to stimulate germination of hard-seeded species. In 8-year-old restoration, soil temperatures during the burns were in excess of 60 °C at a depth of 2 cm and temperatures of 45 °C penetrated to 4 cm. Although fuel loads were higher in rip-line furrows, soil temperatures recorded by the data loggers indicated that both furrows and ridges received sufficient soil heating to stimulate germination of hard-seeded species. The optimal soil depth for heat stimulation of hard-seeded species coincides with the depth from which seedling emergence is most successful (Grant et al., 1996). If dormant seeds are not stimulated to germinate by fire, populations of legume species may decline through reduced recruitment (Auld, 1986; Bradstock and Auld, 1995).

4.4. Post-burn fuel characteristics and vegetation structure

Burning significantly reduced fuel quantities in jarrah forest restoration. Accumulation of litter in 5-year-old restored sites was rapid in the first year post-fire. Although the burn was of low intensity, it scorched much of the lower tree foliage and shrub layer. Abscission of scorched overstorey leaves resulted in a rapid accumulation of fuel in 5-year-old restoration. Within 1 month of burning, fuel loads in 5-year-old restoration had already reached the maximum recommended level for prescription burning in the jarrah forest (8 t ha^{-1}), a level that normally requires 5–10 years to accumulate (Burrows, 1994).

Burns in 8-year-old restoration consumed much of the tree and shrub canopies, resulting in a comparatively reduced post-fire litter-fall. Litter fuel loads remained relatively constant in these sites in the 2 years following burning. Crown scorch led to sudden leaf drop following fire. After this initial leaf-drop, trees undergo a period of canopy replacement during which their leaf abscission is reduced (Birk and Bridges, 1989; McCaw et al., 1996). Litter fuel loads were lower following burning of 8-year-old restored sites due to extensive combustion of the tree canopy, which reduced post-fire litter-fall. High intensity fire greatly decreased fuel connectivity in 8-year-old restoration (from 64 to <14% litter cover). In 5-year-old restoration, however, fuel connectivity significantly increased in both of the 2 years following fire, to levels near that of pre-burn 8-year-old restored sites. This means that 5-year-old sites may be prone to fire within 2 years after burning.

The trash component tended to decrease with time after burning as suspended fuel collapsed. Consistent increases in live standing material post-fire in 8-year-old restored sites was due to plant establishment and resprouting. The development of vegetation structure over the 2 years following burning was significantly affected by the age at which fire was introduced into restored areas. Five-year-old restoration showed an increase in dead plant matter below 2 m in height. Many acacias were scorched and killed by the low intensity fire, however, they were not consumed and remained as trash. In contrast, 8-year-old restoration had almost all vegetation (live and dead) removed by fire. All that remained were tree trunks and thick acacia stems.

The impact of high intensity fire on 8-year-old restoration was to produce, at 2 years post-fire, a vegetation structure similar to that of the jarrah forest 2 years after fire. Following burning in 8-year-old restoration there was an increase in live plant material below 1 m in height after 2 years, reflecting significant post-fire plant establishment. There was also an increase in live material above 2 m within 2 years, due to tree resprouting. Similar results were recorded after high intensity fire in 11–15-year-old jarrah forest restoration (Grant et al., 1997a). In subsequent years, dense establishment of post-fire germinants led to the redevelopment of a mid-storey dominated by acacias. However, increased competition for light and water from the existing overstorey has meant that the biomass of shrubs in the mid-storey is reduced when compared to newly restored sites (C.D. Grant pers. com.). Further monitoring of burnt restored areas is required to assess if high intensity fire will 'recycle' the fire hazard presented by acacias. In unburnt control 8-year-old restored sites the trash layer had started to collapse by the third year of this study. It is possible then, that a two-tiered fuel structure similar to that of the jarrah forest, may develop in the absence of fire, given sufficient time.

Immediately following burning, both restored and jarrah forest sites experienced a significant pulse of litter-fall. The type of vegetation and the severity of fire largely determined the size of the litter-fall pulse. Low intensity prescribed burning in unmined eucalypt forests does not usually affect litter-fall (Raison et al., 1983). Under dry, warm summer conditions, scorch height in the jarrah forest was approximately nine times the height of flames (Burrows, 1997). Therefore, even flame heights as low as 0.5 m have the potential to scorch foliage up to 4.5 m in height. The immature nature of the fuel structure in the 5-year-old sites greatly influenced the fall of leaf material after burning.

The reduction of reproductive material in the litter-fall debris of both restored vegetation communities indicates that the burns have disrupted the reproductive cycles of plant species in these sites. Five-year-old restoration was nearing its peak of understorey seed production at the time the burns occurred (Smith, 2001). The most actively flowering species were in the herb and shrub layers that were scorched by fire. Disruption of reproductive cycles, combined with the subsequent lack of establishment of plants post-burn, indicates that these sites may be impacted to a greater extent, than may have been expected, considering the low intensity of the fires experienced.

4.5. Implications for fire management in jarrah forest restoration

The implications for fire management of jarrah forest restoration in bauxite mined areas in the Darling Ranges of Western Australia, arising from the current study, are summarised below.

1. Fire should be excluded from any sites <8-year-old to ensure that the dominant canopy species are

sufficiently developed to withstand fire. This will also ensure that there is a sufficient topsoil seed bank established to allow fire sensitive shrubs to regenerate and fire tolerant shrubs to become large enough to withstand the impacts of a medium intensity fire.
2. Sites older than 8 years should be burnt at low to medium intensities to reduce fuel loads and to facilitate successional processes. Generally, a fire rotation of at least 10 years and preferably longer is considered most suitable for dry sclerophyll forests (Tolhurst, 1996), although this timeframe will need further investigation within restored jarrah forest sites.
3. Jarrah forest restoration areas to the age of 5 years will act as firebreaks and prevent the majority of fires from becoming or sustaining crown fires.
4. Any prescribed burns should be performed in spring or autumn as this will make the fire easier to control than a summer burn and also encourage greater establishment of understorey species.

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