Short-term impacts of logging on understorey vegetation in a jarrah forest

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Revised manuscript received 5 December 2001

Summary

In 1985, new silvicultural treatments were implemented in jarrah (*Eucalyptus marginata*) forests available for wood production. As part of a scientific investigation into the ecological impacts of two of these treatments, gap cutting and shelterwood cutting, a survey was conducted 4 years after logging to examine the effects of these treatments on understorey vegetation species richness and abundance. Sampling scale was found to be an important factor affecting the results and subsequent interpretation of impacts. At the coupe scale, native plant species richness in unlogged coupe buffers was similar to that in adjacent logged patches. However, the mean number of species per 1 m^2 was 20%–30% higher in the unlogged buffers than the logged patches. At all sampling scales, the abundance (number of individual plants) of native plants was 20%–35% higher in the buffers, but the abundance of introduced (weed) species was significantly higher in the logged patches. The abundance of weeds, which are mostly annual grasses and short-lived herbs, is likely to diminish with time. The time to recovery of native species abundance and the ecological significance of this is uncertain. Given the reported low seedling regeneration rate and limited dispersal capacity of many woody shrubs and perennial herbs, they are unlikely to return to pre-logging levels in the medium term. We attribute the reduction in the abundance of native plants mainly to mechanical soil disturbance, which ranged from 60% to 80% of the area of logged coupes, physical damage to the vegetation associated with logging and to intense heating of the topsoil during the post-logging silvicultural burn. Recommendations are made for reducing the negative impacts of logging operations on the understorey.

Keywords: logging; environmental impact; understorey; vegetation types; soil disturbance; stand development; silvicultural systems; regeneration; *Eucalyptus marginata;* Western Australia

Introduction

Modern forest management policies are formulated around the concept of ecologically sustainable development (ESD) and from that, ecologically sustainable forest management (ESFM). According to the Ecologically Sustainable Development Working Groups (1991), this implies optimising the material and non-material, social and economic benefits that forests can provide to the community with the goals of maintaining the functional basis of the forest, biodiversity, and the options for future generations.

Western Australian government policy and community expectation require that forest management, including timber extraction, should be ecologically sustainable and should optimise benefits to the community (CALM 1992). In order to implement such policies, forest management agencies recognise the need for basic ongoing research to gain scientific knowledge of forest ecosystem processes and how these are affected by forestry operations. Management agencies also recognise the need to monitor the implementation and impact of these operations to gauge the success of their policies in regard to achieving ESFM. Meaningful monitoring protocols can only be designed and implemented with a firm understanding of these processes and of key ecosystem elements that should be monitored.

While there are a number of unpublished reports, there are few publications that report on the effects of logging on vegetation in Australian eucalypt forests (e.g. Cremer and Mount 1965; Hickey 1994; Ough and Murphy 1996; Murphy and Ough 1997). These studies were post-treatment surveys rather than controlled experiments. There is little published literature specifically on the ecological effects of logging on Western Australian forests and there are no published accounts of the impact of logging on jarrah forest vegetation assemblages. The Kingston Project, described in detail by Burrows *et al.* (1994), is a multi-disciplinary scientific investigation into the short-term impact of logging on jarrah forest ecosystems, including the understorey vegetation, which is important in its own right and as habitat for a diverse suite of fauna. Mosses and lichens were investigated in a separate study so are not dealt with here. The Kingston Project specifically examined impacts in response to current silvicultural prescriptions, which were introduced in the 1980s (Bradshaw 1985; CALM 1995). The short-term impacts of logging on understorey vegetation were examined using both an experimental approach (before, after, control, impact) incorporating a series of 30 m × 30 m (900 m^2) plots (Burrows *et al.* 1994) and by a survey conducted 4 years after logging and burning using a series of 1 m × 1 m sample plots. This paper reports the findings of the survey.

Methods

Study area

The survey was conducted in jarrah forest about 25 km north-east of Manjimup in the south-west of Western Australia (34°15′S, 119°09′E). Sample sites were located in Kingston, Winnejup and Warrup State forest blocks, each about 3500 ha. The region has a Mediterranean-type climate with cool wet winters and warm dry summers. Annual average rainfall is about 800 mm and its strongly seasonal nature is reflected in the ratio of winter (April to October) to summer (November to March) rainfall, which is about 6:1. Rainfall is reliable, with the variability being only about ±15% of the mean. The annual potential evaporation is about 1500 mm and the mean monthly maximum temperature ranges from 16°C to 31°C.

The Kingston Project study area in which the survey was conducted encompasses about 11 000 ha of State forest and lies within the Darling Botanical District with the predominant vegetation on the ridges and uplands being an open forest of jarrah (*Eucalyptus marginata*) and marri (*Corymbia calophylla*). The topography is a relatively simple system of gently undulating plateau tops, low lateritic ridges, broad valley floors, creeks and rivers. The uplands and ridges are characterised by sandy yellow/brown gravels with occasional boulders and sheets of laterite, while yellow/brown podzolic soils occur along drainage lines. Grey/brown clay/loams are commonly associated with broad valley floors with sands occurring around the margins of swamps. Churchward (1999) provides detailed descriptions of the physiography, geology and soils of the area. The mature upland forest has a top height of about 25 m, a basal area of 25–35 m^2 ha^{-1} and an overstorey canopy cover of about 40%. The understorey vegetation on these sites is low (mostly <1.5 m high) and relatively open (<60% cover) and comprises a diverse range of perennial herbs, woody shrubs and small trees such as *Banksia grandis*, *Persoonia longifolia*, *Macrozamia reidlei*, *Xanthorrhoea preissii*, *Bossiaea ornata, Hakea lissocarpha* and *Leucopogon capitellatus*. In treeless drainage lines on shallow soils, *Hakea prostrata*, *H. varia* and *Acacia saligna* sometimes form tall open thickets to 3 m. *Banksia littoralis, Hakea oleifolia* and *Eucalyptus rudis* are common along creek lines, valley floors and seasonally swampy areas. These habitats are excluded from logging.

The Mediterranean-type climate, together with accumulations of flammable fuel, makes the region prone to fire. Fire history prior to the 1930s is uncertain, but records kept since 1938 reveal a mosaic of infrequent but often intense summer wildfires and low- to moderate-intensity prescribed burns at 7–10 year intervals in both spring and autumn. Prior to the commencement of the Kingston Project in 1994, the area was last burnt in autumn 1986. Most of the study area was cut-over during the 1940s and 1960s, although some patches show no record or evidence of having been logged. Based on the evidence of tree stumps remaining and the structure of the existing forest, the early logging was light and selective, with only the trees of best form being utilised.

Current jarrah forest silvicultural systems

In jarrah forest available for timber harvesting, one of three silvicultural systems will usually be applied to a patch of forest, depending on the existing stand structure and density of regeneration (Bradshaw 1985). These are:

1. Thinning to promote growth on retained jarrah and marri trees.
2. Removing the overstorey (creating gaps) to release and promote the growth of jarrah and marri advance growth existing as seedlings, ground coppice (advanced seedlings with well developed lignotubers) and small saplings. Maximum gap size is 10 ha, with most gaps being 4–7 ha.
3. Cutting to retain a shelterwood to establish regeneration from seed where advance growth does not exist in sufficient density. Seedlings will be encouraged to establish and develop into ground coppice by reducing competition from the overstorey. A forest canopy is maintained (basal area about 13 m^2 ha^{-1}) to provide a continuity of forest values until the ground coppice is developed and capable of responding to release following canopy removal.

The choice of silvicultural treatment applied is determined following a ground survey of the extent and nature of existing regeneration. Habitat trees and habitat logs are identified and marked for retention, permanent buffers (unlogged areas) are retained along roads and streams and unlogged buffers are retained between coupes (coupe buffers). Coupe buffers will or may be logged in the next cutting cycle, some 15–30 years in the future. In the Kingston Project area about 17% of the area was cut to create gaps, 16% was cut to retain shelterwood and 67% of the area was not logged in this cutting cycle.

Some manual/machine culling of non-commercial trees, including jarrah, marri, sheoak (*Allocasuarina fraseriana*) and bull banksia (*Banksia grandis*) that may compete with regeneration of commercial tree species or retained crop trees is carried out either during or after logging. As well, understorey competition is mechanically reduced and the soil is disturbed using a rubber-tyred machine to form a receptive seedbed for tree species (jarrah and marri). Logged areas are normally given a prescribed burn after logging to stimulate seed fall, to create receptive seedbeds, to remove logging debris and to reduce the fuel levels available during a wildfire. About one-third of the study area was logged during 1995/96 and the post-logging prescribed burn was carried out in late spring 1996. Standard silvicultural treatments and logging practices were applied (Burrows *et al*. 1994). In addition to silvicultural prescriptions, logging operations are guided by a documented code of practice (CALM 1997) and by the manual of logging specifications (CALM 1993).

Study design and data collection

A survey of the number of understorey species (richness) and abundance (number of individual plants) was carried out 4 years after logging and burning (in spring 2000) using a series of 1 m^2 sample quadrats located in each of five treatments described in Table 1. An advantage of space-for-time surveys is that information about disturbance effects can be gathered relatively quickly. However, a limitation of this technique is that it assumes

Table 1. Management history of treatment sites that were surveyed for the effects of silvicultural treatments on plant species richness and abundance in a jarrah forest.

Management history of treatment sites
External control: Lightly cut over 1940s and 1960s, last burnt autumn 1986
Shelterwood cut: Lightly cut over 1940s and 1960s, logged 1995/96, last burnt spring 1996
Shelterwood coupe buffer: Lightly cut over 1940s and 1960s, last burnt spring 1996
Gap cut: Lightly cut over 1940s and 1960s, logged 1995/96, last burnt spring 1996
Gap coupe buffer : Lightly cut over 1940s and 1960s, last burnt spring 1996

differences between treatment and control sites are due to the treatments and not to other factors such as site variation. The null hypothesis under test was that there is no significant difference in terms of a) plant species richness and b) abundance, between the silvicultural treatments and the controls (adjacent coupe buffers and external controls). The two silvicultural treatments (cutting to create gaps and cutting to retain shelterwood) are described above. Prior to establishing the 1 m × 1 m sampling quadrats, sample lines were located within each treatment. In logged treatments, sample lines commenced 20 m from the edge of the logging boundary and continued 100 m into the logged area. Sample lines were also placed in similar sites (with respect to vegetation type, soils and topography) immediately adjacent to the treatments in the coupe buffers and in an external control treatment several kilometres from the logged treatments. The external control site was last burnt in 1986 so represented the least disturbed forest in the study area (Table 1). At 10 m intervals along each 100 m sample line, a quadrat was located on the line and 5 m either side of the line giving a total of 30 quadrats per line and 150 quadrats per treatment. The species and the number of individual plants of each species were recorded for each quadrat. For this forest type, the understorey was considered to be all vegetation up to 2 m above ground.

Prior to the survey, a list of vascular plant species was prepared for the area by collecting specimens throughout the year; voucher specimens were lodged with the Western Australian Herbarium. Species were grouped into eight life forms/habits, or guilds, according to Paczkowska and Chapman (2000) (Appendix 1). Except for the ‘weed’ group, all other taxa are native to the study area. Most weed species are either short-lived herbs or annual grasses. All taxa were allocated to one of the following guilds:

1. Tree — a single stemmed woody plant >5 m tall when mature with a distinct main axis;
2. Shrub — a woody plant <5 m tall when mature either without a distinct main axis, or with branches persisting on the main axis;
3. Perennial Herb — a non-woody plant with a lifespan >2 years;
4. Short-lived Herb — a non-woody plant with a lifespan ≤2 years;
5. Grass;
6. Sedge;
7. Fern;
8. Weed — introduced or alien species.

Species were also grouped according to their main method of regenerating after fire (post-fire regeneration strategy), which was either determined from field observations or from Paczkowska and Chapman (2000). Three groups were identified for the purposes of this study:

1. Resprouts from a lignotuber (woody rootstock) or epicormic shoots;
2. Resprouts from fleshy underground organ including roots, bulbs, corms, rhizomes or tubers;
3. Depends mainly on seed stored in the soil or in the canopy.

The extent of mechanical soil disturbance associated with the various treatments was assessed as part of another study in spring 1999, about 12 months prior to this survey. This involved recording the condition of the soil at 2 m intervals along a series of 30 m line transects located within the treatments. At each sample point, the soil surface was ascribed to one of the categories shown in Table 7.

Data analysis

Data were analysed for each of the treatments (Table 1) and life forms (guilds) described above and at three sampling scales derived by the following combinations of the 1 m^2 quadrats:

- Total richness and abundance recorded in each treatment for all 150 quadrats combined (150 m^2 sampling scale);
- Mean richness and abundance per transect line calculated from the 5 transects, or 30 quadrats, per treatment (30 m^2 sampling scale); and
- Mean richness and abundance per quadrat calculated from 150 quadrats per treatment (1 m^2 sampling scale).

Comparisons were made (difference between two means) for a) a silvicultural treatment and adjacent coupe buffer, b) a silvicultural treatment and external control, and c) external control and coupe buffer. When comparing means, the null hypothesis (that there was no significant difference between the means) was tested using a two-tailed *t*-test ($\alpha = 0.05$). Statistical power was diminished when analysing data at the 30 m^2 sampling scale where $n = 5$, compared with $n = 150$ at the 1 m^2 sampling scale. We chose to fix α at 0.05, recognising that this increased the chance of making a Type II error (accepting a hypothesis that is in fact false).

Results

A total of 175 understorey taxa (including 14 introduced weed species) representing 48 families were recorded across all treatments from 750 × 1 m^2 quadrats (see Appendix 1). Eleven taxa were not identified to species level, five of these being orchids for which the leaf base but no flower was evident at the time of assessment. Following disturbance by logging and fire, 32% of species had regenerated by seed and 60% had resprouted from underground organs or epicormic shoots. The regeneration method of the remainder (8%) is unknown (Appendix 1). The most commonly represented families were Papilionaceae, Mimosaceae, Proteaceae, Asteraceae, Orchidaceae and Apiaceae (see Appendix 1). The proportion of all taxa by life form category is shown in Figure 1.

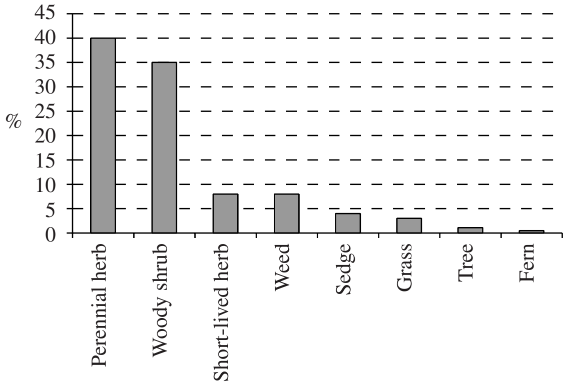

Figure 1. Proportion of plant species recorded in the study area by life form categories. The exotic species (weeds) were mostly annual grasses and short-lived herbs.

Table 2. Total number of species recorded in shelterwood-cut jarrah forest patch and adjacent coupe buffer assessed four years after disturbance

Life form (guild)	Common to both treatments	Recorded only in logged patch	Recorded only in coupe buffer
Tree	2	0	0
Woody shrub	33	3	4
Perennial herb	44	10	7
Short-lived herb	6	4	1
Grass	4	0	0
Sedge	4	0	0
Fern	1	0	0
Weed	9	2	0

Species richness — shelterwood cut compared with adjacent coupe buffer

Four years after the silvicultural burn, the total number of native plant species recorded in the 150 (1 m^2) quadrats per treatment was similar in forest patches cut to shelterwood and in adjacent coupe buffers, with about 90% of species common to both areas (Table 2). The long unburnt external control had (marginally) the lowest level of species richness (Table 3). At the guild level there were only slight variations in species richness between the logged treatment and the adjacent coupe buffer. For example, there was one more woody shrub species recorded in the buffer, but three more perennial herbs and three more short-lived herbs recorded in the logged forest. The total number of grass and sedge species was the same for both treatments, but both contained about double the number of weed species as the external control (Table 3).

At the 30 m^2 sampling scale, mean native plant species richness was about 4% higher in the adjacent coupe buffer (not statistically significant) and the external control had the lowest level of native species richness measured at this resolution (not statistically significant) (Table 3). Mean woody shrub species richness was virtually the same for the logged treatment and the adjacent coupe buffer, but the richness of perennial herbs was about 14% higher in the buffer (not statistically significant). The mean species richness of short-lived herbs was almost 60% higher in the logged forest (statistically significant), but there was no significant difference between mean species richness of grasses and sedges.

At this sampling scale (30 m^2), the mean number of weed species was considerably higher (statistically significant) in the logged forest (6.2 species per 30 m^2) compared with the buffer (3.4 species per 30 m^2) and the external control (1.4 species per 30 m^2) (Table 3).

In contrast to data examined at 150 m^2 and 30 m^2 sampling scales, mean native plant species richness measured at the 1 m^2 sampling scale was 23% higher in the adjacent coupe buffer compared with the logged forest, which was statistically significant (Table 3). This pattern was similar for woody shrubs (21% higher in the buffer) and perennial herbs (31% higher in the buffer). There was no significant difference in mean grass species richness but the mean species richness of short-lived herbs was 30% higher in the logged forest and weeds were 87% higher in the logged forest. There was no significant difference in mean native plant species richness (m^{-2}) between the external buffer and the logged treatment, but species richness in the buffer was significantly higher than the external control, which had not been burnt since 1986.

Species richness — gap cut compared with adjacent coupe buffer

As with the shelterwood cut, the total native plant species richness at the 150 m^2 sampling scale was similar for both logged and coupe buffer sites, which were about 12% higher than the external control; the difference was mainly due to lower numbers of woody shrubs and short-lived herbs in the external control (Table 3). About 86% of native plant species were common to both treatments (logged and buffer), a proportion similar to the shelterwood sites. However, there was variation at the guild level (Table 4). The number of woody shrub species was about 8% higher in the logged treatment, but the number of perennial herbs was about 10% lower, neither of which was statistically significant. Similarly, the number of sedge species was lower in the logged area (three compared with five in the buffer). As with the shelterwood sites, the number of short-lived herbs and weed species was higher in the logged area, but the number of grass species was the same.

At the 30 m^2 sampling scale, the mean number of native species was 8% higher in the coupe buffer (not statistically significant). Mean species richness of woody shrubs was similar, but the mean number of perennial herb species was higher in the buffer (not statistically significant) (Table 3). Mean grass species richness was the same for both treatments, but short-lived herbs and weeds were higher in the logged area (not statistically significant). There were too few sedges and ferns in both areas to make statistical comparisons at this sampling scale.

The mean number of native species recorded per quadrat (the 1 m^2 sampling scale) was about 33% higher (statistically significant) in the buffer compared with the logged patch of forest (Table 3). At the guild level, the mean species richness of perennial herbs was about 50% higher in the buffer (statistically significant) and the number of woody shrub species was about 40% higher in the buffer (statistically significant). The number of grass species was also significantly higher in buffer, but there were significantly more short-lived herb and weed species in the logged forest

Table 3. Understorey plant species richness (number of species) by life form and treatment for three sampling scales, where A = total number of species recorded in 150 × 1 m^2 quadrats for each treatment, B = mean number of species per 30 m^2 (n = 5 lines each of 30 × 1 m^2 quadrats per treatment) and C = mean number of species per 1 m^2 (n = 150 for each treatment). Standard errors in parentheses.

† Indicates mean is significantly different from the adjacent coupe buffer at the 0.05 level.
◆ Indicates mean is significantly different from the external control at the 0.05 level.

Life form (guild) and sampling scale		External control site Unlogged, 14 years unburnt	Shelterwood sites		Gap sites	
			Logged, burnt	Adjacent coupe buffer, burnt	Logged, burnt	Adjacent coupe buffer, burnt
Tree (< 2m)	A.	2	2	2	2	2
	B.	2	2	2	2	2
	C.	0.63 (0.06)	0.31†◆ (0.04)	0.65 (0.05)	0.4†◆ (0.05)	0.7 (0.05)
Woody shrub	A.	33	36	37	41	38
	B.	17.2 (0.44)	18.4 (1.4)	18.0 (2.4)	20.4 (3.31)	19.8 (1.88)
	C.	4.1 (0.19)	3.2†◆ (0.15)	3.9 (0.14)	4.0† (0.18)	5.7◆ (0.19)
Perennial herb	A.	53	54	51	49	54
	B.	26.0 (2.14)	24.4 (1.62)	27.8 (2.01)	20.4 (3.89)	25.6 (1.77)
	C.	4.6 (0.22)	3.5†◆ (0.18)	5.1◆ (0.17)	3.2†◆ (0.18)	4.9 (0.19)
Short-lived herb	A.	6	10	7	13	11
	B.	2.8 (0.86)	6.0†◆ (0.31)	4.6◆ (0.50)	6.2◆ (0.48)	4.4◆ (1.16)
	C.	0.2 (0.05)	1.8†◆ (0.12)	1.2◆ (0.09)	1.6†◆ (0.09)	1.0◆ (0.09)
Grass	A.	4	4	4	4	4
	B.	2.4 (0.24)	3.0 (0.31)	3.0 (0.30)	2.8 (0.20)	2.8 (0.48)
	C.	0.7 (0.06)	0.8 (0.06)	1.1◆ (0.06)	0.7† (0.05)	1.0◆ (0.06)
Sedge	A.	3	3	3	3	5
	B.	1.0 (0.44)	1 (0.43)	1.4 (0.44)	0.8 (0.48)	2.2 (0.58)
Fern	A.	1	1	1	1	1
Weed	A.	5	11	9	10	8
	B.	1.4 (0.40)	6.2†◆ (1.11)	3.4◆ (1.06)	5.4◆ (1.50)	3.0◆ (1.04)
	C.	0.3 (0.06)	1.7◆ (0.10)	0.8◆ (0.09)	1.2◆ (0.11)	0.5◆ (0.06)
All native species	A.	102	107	105	114	115
	B.	51.4 (2.94)	54.8 (3.11)	56.8 (3.92)	52.6 (7.27)	56.8 (3.46)
	C.	10.1 (0.33)	9.6† (0.31)	11.9◆ (0.27)	10.1† (0.33)	13.3◆ (0.27)

Table 4. Total number of plant species recorded in gap-cut forest patches and adjacent coupe buffers assessed four years after disturbance

Life form (guild)	Common to both treatments	Only in logged patches	Only in coupe buffer
Tree	2	0	0
Woody shrub	35	6	3
Perennial herb	43	6	11
Short-lived herb	10	3	1
Grass	4	0	0
Sedge	3	0	2
Fern	1	0	0
Weed	8	2	0

(Table 3). The coupe buffers adjacent to the gap-cut sites had the highest number of native species per quadrat of all the treatments.

Plant abundance – Shelterwood cut compared with adjacent coupe buffer

The total number of individual native plants (abundance irrespective of species) recorded in the adjacent coupe buffer was 35% higher than in the logged forest patches (Table 5). By guilds, however, there was considerable variation between the two treatments. The abundance of woody shrubs was 16% higher in the buffer (not statistically significant at either the 30 m^2 or the 1 m^2 sampling scales). However, the abundance of perennial herbs was 140% higher in the buffer, which was highly significant at both the 30 m^2 and 1 m^2 sampling scales (Table 5). The abundance of sedges was nine times higher in the buffer (statistically significant at the 1 m^2 sampling scale), but the abundance of short-lived herbs and weeds was several times higher in the logged patches (statistically significant at both the 30 m^2 and 1 m^2 sampling scales). The abundance of the fern *Pteridium esculentum* was also significantly higher in the logged patches. The total abundance of native species in the external control was similar to the logged forest, but this was due to the high abundance of short-lived herbs and grasses in the logged forest. The abundance of woody shrubs and perennial herbs was highest in the external control and the abundance of weeds was lowest (Table 5).

Plant abundance — gap cut compared with adjacent coupe buffer

The total abundance (number of plants) of all native plant species in forest patches cut to gaps was about 21% lower than in the adjacent coupe buffer (Table 5) but there was considerable variation between the guilds. The mean abundances of woody shrubs and perennial herbs were about 42% and 82% higher, respectively, in the adjacent buffer compared with the logged forest (both statistically significant). The abundance of sedges and grasses was also significantly higher in the buffer. In the

Table 5. Understorey plant species abundance (number of plants) by life form and treatment for three sampling scales, where A = total number of species recorded in 150 × 1 m^2 quadrats for each treatment, B = mean number of species per 30 m^2 (n = 5 lines each of 30 × 1 m^2 quadrats per treatment) and C = mean number of species per 1 m^2 (n = 150 for each treatment). Standard errors in parentheses.

† Indicates mean is significantly different from the adjacent coupe buffer at the 0.05 level.

◆ Indicates mean is significantly different from the external control at the 0.05 level.

Life form (guild) and sampling scale		External control site Unlogged, 14 years unburnt	Shelterwood sites		Gap sites	
			Logged, burnt	Adjacent coupe buffer	Logged, burnt	Adjacent coupe buffer
Tree	A.	172	56	141	85	144
	B.	34.5 (7.69)	11.1†◆ (2.03)	28.2 (3.76)	17.0†◆ (3.67)	28.8 (4.53)
	C.	1.15 (0.14)	0.37†◆ (0.05)	0.92 (0.06)	0.57†◆ (0.06)	0.96 (0.10)
Woody shrub	A.	1327	937	1086	1308	1857
	B.	265 (68)	187 (40)	217 (73)	261 (39)	371 (45)
	C.	8.9 (0.55)	6.2†◆ (0.39)	7.2◆ (0.32)	8.7† (0.49)	12.4◆ (0.53)
Perennial herb	A.	2488	1431	3442	1584	2889
	B.	497 (63)	286†◆ (27)	688◆ (121)	317†◆ (77.3)	578 (65.7)
	C.	16.6 (1.58)	9.5†◆ (0.74)	22.9◆ (1.13)	10.5†◆ (0.70)	19.3 (0.79)
Short-lived herb	A.	91	1515	660	1446	616
	B.	18 (12)	303†◆ (58)	132◆ (30)	289†◆ (104.1)	123◆ (61.0)
	C.	0.6 (0.17)	10.1†◆ (1.10)	4.4◆ (0.45)	9.6†◆ (1.17)	4.1◆ (0.64)
Grass	A.	240	324	418	258	507
	B.	48 (8.4)	65 (23)	84 (18)	52† (7.4)	101 (14.0)
	C.	1.6 (0.20)	2.2 (0.24)	2.8 (0.26)	1.7† (0.17)	3.4◆ (0.50)
Sedge	A.	41	12	108	8	180
	B.	8.1 (2.7)	2.4†◆ (1.1)	21.6◆ (4.1)	1.6†◆ (1.1)	36.0◆ (10.8)
Fern	A.	19	68	7	25	51
Weed	A.	150	2212	594	1437	340
	B.	30 (50)	442†◆ (96)	119◆ (45)	287†◆ (182)	68 (50)
	C.	1.0 (0.25)	14.8†◆ (1.99)	3.9◆ (0.87)	9.6†◆ (1.75)	2.3◆ (0.39)
All native species	A.	4378	4343	5862	4714	5689
	B.	876 (86)	869† (87)	1172◆ (70)	943† (151)	1138◆ (68)
	C.	29.2 (1.74)	28.9† (1.44)	39.1◆ (1.35)	31.4† (1.54)	37.9◆ (1.53)

Table 6. The proportion of plant species (number of plants per species) that declined in the logged treatments compared with coupe buffers by life form and regeneration strategy

Life form and regeneration strategy (guild)	Proportion of all native species recorded (%)	Proportion of all species that declined (%)	Declined species as a proportion of all native species (%)	Proportion of species that declined in that guild (%)
Woody shrub	35	37	11	28
Perennial herb	40	54	15	35
Sedges	4	7	2	50
Grasses	3	2	1	20
Seeders	24	21	6	24
Lignotubers	22	24	7	28
Corms, bulbs, tubers, rhizomes	48	55	15	29

logged patches the abundances of short-lived herbs and weeds were 57% and 322%, respectively, higher than in the buffer. While there was a significant difference in the abundance of tree species (higher in the buffer), this was to be expected as a proportion of the ground coppice present prior to logging had developed through to the sapling stage following logging and, being taller than 2 m, was not recorded.

About one-third of all native plant species were more abundant in the coupe buffer than in the logged patches (both gap and shelterwood logging treatments), so are assumed to have declined as a result of logging. Those guilds that were least abundant in both the gap-cut and shelterwood-cut (logged) areas were sedges (50% of taxa were less abundant), perennial herbs (35% of taxa were less abundant) and woody shrubs (28% of taxa were less abundant) (Table 6). Species dependent on corms, bulbs, tubers and rhizomes comprised 48% of all native taxa, but comprised 55% of taxa that were less abundant in the logged patches. (Table 6). Woody shrubs, perennial herbs and sedges (native species) that were more abundant in the buffers include *Acacia pulchella*, *Banksia grandis*, *Bossiaea linophylla*, *Caladenia flava*, *C. reptans*, *Clematis pubescens*, *Craspedia pleiocephala*,

Table 7. Proportion (%) of each treatment affected by various soil disturbance classes. The gap-cut and shelterwood-cut treatments were measured three years after logging and silvicultural burning. The coupe buffer was burnt three years prior to measurement, but was not logged. The external control was measured at the same time but was last burnt 14 years prior to measurement.

Disturbance class	External control (n = 1440)	Coupe buffer (n = 1300)	Gap cut (n = 1499)	Shelterwood cut (n = 720)
Undisturbed	82	74	18	7
Old mechanical disturbance	13	6	6	0
Old landing	1	0	0	0
New mechanical disturbance	0	10	53	68
New landing	0	0	5	1
Ashbed	0	7	20	19
Animal digging	4	3	3	5

Table 8. Proportion (%) of bare soil and litter cover and mean litter depth. The gap-cut and shelterwood-cut treatments were measured three years after logging and silvicultural burning. The coupe buffer was burnt three years prior to measurement, but was not logged. The external control was measured at the same time but was last burnt 14 years prior to measurement.

Cover class	External control	Coupe buffer	Gap cut	Shelterwood cut
Bare soil	11	28	76	74
Litter cover	85	70	22	25
Other (e.g. rock, log)	4	2	2	1
Mean litter depth (mm)	14.1	7.3	1.7	1.6

Cyrtostylis huegelii, *Desmocladus fascicularis*, *Drosera huegelii*, *D. menziesii*, *Geranium solanaderi*, *Gompholobium marginatum*, *Hakea lissocarpha*, *Hardenbergia comptoniana*, *Hibbertia amplexicaulis*, *H. racemosa*, *Lagenophora huegelii*, *Leucopogon capitellatus*, *L. propinquis*, *Lepidosperma leptostachyum*, *Leptomeria cunninghamii*, *Lomandra drummondii*, *L. integra*, *L. sericea*, Orchid sp. 1, Orchid sp. 2, *Pelargonium littorale*, *Pentapeltis silvatica*, *Ranunculus colonorum*, *Sollya heterophylla*, *Sphaerolobium medium*, *Tetraria capillaris*, *T. octandra*, *Tetrarrhena laevis*, *Tetratheca hispidulus*, *Thysanotus pattersonii*, *Trymalium ledifolium*, *Veronica calycina*, *Xanthosia atkinsonia*, *X. candida* and *X. heugelii*.

Species that were generally more abundant in the logged areas were mainly weeds or short-lived native herbs and include *Aira caryophyllea*, *Amperea ericoides*, *Centaurium erythraea*, *Conyza bonariensis*, *Daucus glochidiatus*, *Euchiton collinus*, *Galium murale*, *Hypochaeris glabra*, *Kennedia prostrata*, *Lotus suaveolens*, *Luzula meridionalis*, *Millotia tenuifolia*, *Opercularia hispidula*, *Oxalis corniculata*, *Phyllanthus calcinus*, *Pseudognaphalium luteo-album*, *Trachymene pilosa*, *Trifolium campestre* and *Vulpia myuros*.

In summary, four years after logging and burning, the abundance of perennial herbs, sedges, grasses and woody shrubs was substantially lower in the logged patches than in the coupe buffers and the external control. This is in contrast to short-lived herbs and weeds, which were more abundant in the logged patches. There was no consistent trend for the only fern recorded (*Pteridium esculentum*).

Soil disturbance

The extent of soil disturbance by disturbance classes and soil cover is summarised in Tables 7 and 8 for each treatment. As expected, the level of soil disturbance resulting from logging activities was highest in the logged patches and least in the external control and the coupe buffer. Machines passing through the buffers lightly disturbed about 10% of the buffer area. Cover and depth of litter was greatest in the least disturbed treatment, the external control (Table 8), reflecting the long absence (14 years) of fire. Ashbeds, identified by visual indicators of the soil being exposed to high temperatures for long periods, such as changes to the soil structure and colour, were most frequent in the logged areas, as expected (Table 8). Animal diggings, which were recorded at similar levels across the treatments, were attributed mostly to woylies (*Bettongia penicillata*) and quenda (*Isoodon obesulus*).

Discussion

At the coupe scale, the number of native plant species (richness) was similar in logged coupes and adjacent coupe buffers when surveyed 4 years after logging and burning. However, the abundance (number of individual plants) of native perennial herbs, sedges and woody shrubs was significantly lower in the logged coupes, but the abundance of short-lived herbs and introduced weeds was higher. Within the life form guilds, there were no strong patterns of decline associated with regeneration strategy, although species regenerating mainly from corms, bulbs, tubers and rhizomes (non-lignotuberous species) were over-represented in the group of species that had declined by more than about 10% (Table 6). The most likely possible causes of the reduction in the abundance of these taxa following logging are: a) soil disturbance and damage; b) physical damage to the vegetation; c) silvicultural burning; d) post-logging herbivory; and e) a combination of these factors. While it is possible that a combination of factors contributed to the reduction, we believe that soil disturbance and physical damage to the vegetation caused during and after logging operations is the primary cause, with localised super-heating of the soil during the silvicultural burn also contributing.

The area of forest affected by mechanical soil disturbance was about 60% in patches cut to create gaps and about 70% in patches cut to shelterwood. The soil is mechanically disturbed by heavy machinery during felling, log extraction and stockpiling operations. It is also deliberately disturbed during, or soon after logging to prepare a seed bed suitable for jarrah regeneration and to reduce understorey competition to jarrah seedlings and coppice growth. Lignotubers, roots and subterranean storage organs may be sensitive to physical damage (crushing) or disturbance (Ough and Ross 1992). Corms, rhizomes, bulbs and tubers are fleshy underground storage organs that are relatively close to the soil surface (3–8 cm depth) (Pate and Dixon 1982) and so are particularly vulnerable to mechanical damage (compaction, crushing), disturbance or desiccation due to exposure. We also observed that lignotuberous tree and shrub species were killed when plants were up-rooted, crushed, or had root systems severely disturbed by machines. The low abundance of some obligate seed species post-logging may be associated with mechanical compaction and mixing of soil profiles, or harvesting of seed by ants and other animals. However, disturbance clearly favoured the regeneration of short-lived herbs and some introduced species. The fate of seed banks during logging operations deserves further study.

Localised but intense soil heating during the silvicultural burn probably contributed to the decline in native plant abundance. The amount and nature of debris remaining after logging operations presents a vastly different fuel complex than that which accumulates 'naturally' in an unlogged forest. There is ample evidence of the resilience of jarrah forest plant communities to fire in forests carrying 'natural' fuels as distinct from the 'unnaturally' heavy fuels resulting from logging (e.g. Gardner 1957; Christensen and Kimber 1975; Bell *et al*. 1989). Other workers have shown that plants that depend on vegetative propagules, or soil-stored seed, for regeneration following fire are rarely affected by fires of low to moderate intensity because the soil serves to insulate these from lethal temperature regimes (Beadle 1940; Gill 1981; Burrows 1999). However, extremely high soil temperatures have been recorded beneath very heavy fuels, including burning piles of logs, which can cause localised sterilisation of the soil (Beadle 1940; Cromer and Vines 1966; Tunstall *et al*. 1976) and death of propagules. In this study, about 20% of the soil surface was affected by high temperatures (as indicated by ashbed) in the logged treatments. While not specifically measured, we observed that woody shrubs, especially those with lignotubers and subterranean storage organs, were greatly reduced on ashbeds, which were often dominated by mosses and annuals.

Post-logging grazing by native herbivores such as the western grey kangaroo (*Macropus fuliginosus*), or harvesting of seed by ants (Majer and Abbott 1989) may have had an impact, but the reduction in plant density was across a wide range of taxa, many of which are either toxic or are not palatable. The control sites (coupe buffer and external control) would have also been subjected to grazing, but it is not known whether grazing is likely to be more or less intense in recently logged forests. Because of the large scale of the study (treatments carried out in patches over 11 000 ha of forest) we would expect herbivory to be dispersed over this area and therefore not have high impact locally.

Murphy and Ough (1997) conducted a space-for-time survey in mountain ash (*Eucalyptus regnans*) forests in Victoria to investigate the regenerative strategies of understorey flora following logging (clearfelling). In an unbalanced design, they used thirty 1 m × 1 m sample plots (total 30 m^2) in each of three logging coupes and four 30 m × 30 m plots (total 3600 m^2) in each adjacent unlogged forest to collect data. While acknowledging the limitations of the study, they reported a different understorey composition between one-year-old post-logging regeneration and unlogged patches of forest. They also found that in the clearfelled forest, almost all species regenerated from seed (seeders), whereas in the unlogged forest, species that regenerated vegetatively (resprouters) were more common. As discussed above, we found no such bias in this study. Murphy and Ough (1997) concluded that the likelihood of the re-establishment of resprouters was minimal within a 50–80 year rotation period. While not reporting any soil disturbance data, they suggested that changes in floristic composition following logging might have been due to soil disturbance, which may adversely affect the survival of soil-stored seed and of subterranean regenerative organs. Harris (1989), working in *E. regnans* forest in Victoria, reported that all shrub species present before logging (clearfelling) were present when assessed 3 years after logging and, as reported by Murphy and Ough (1997), most species regenerated from seed. Harris (1989) and Ough and Murphy (1996) found that logging adversely affected tree-ferns and epiphytic ferns, taxa that do not occur in the more xerophytic jarrah forest.

The appropriate scale at which to examine and interpret ecosystem responses and 'scaling up' from data collected from small quadrats to interpreting responses over larger areas, has been a vexed issue for landscape ecologists (e.g. Dodson *et al*. 1998; Turner 1998). Spatial and temporal scales are also critical when assessing or evaluating the ecological sustainability of forest management. Clearly, not all species can be expected to be present on every square metre all of the time. This was borne out in this study, where measured variations in plant species richness particularly, and the statistical significance of these variations, depended on the spatial resolution (sampling scale) of the data set. Thus, at the 1 m^2 level, there were significant differences in plant species richness between logged treatments and adjacent buffers, but pooling the quadrats at the 30 m^2 and 150 m^2 levels, there were no significant differences, or differences in total values in the case of the latter. Nonetheless, there was a significant difference in native plant abundance at all levels of resolution, with significantly lower abundances recorded in logged patches. These findings are of course, linked. A significant reduction in plant abundance as a result of logging is likely to reflect a loss of species when examined at a small sampling scale such as the number of species per square metre. Without the benefit of data gathered, analysed and reported at larger sampling scales, this could lead to a misinterpretation of the disturbance effects.

Four key questions relevant to ESFM have emerged from this study:

1. At what spatial scale should plant diversity be measured when assessing the impacts of logging with respect to ESFM objectives?
2. When will logged patches return to plant abundance levels similar to that of unlogged forests?
3. How long will weeds persist at high levels of abundance in logged patches?
4. What are the longer term ecological consequences of reductions in the abundance of native taxa and increases in the abundance of weeds?

With respect to spatial scales for assessing ESFM, we suggest that forest activities should not cause extinctions, or declines to irretrievably low levels, at the scale of the logging coupe, which, for jarrah forest patches cut to gaps, is <10 ha. Questions about time scales for post-logging recovery of plant species richness and abundance, and the persistence of weed species, are not readily resolved because the reproductive biology and dispersal capacity of many plant species is poorly known. In studying the seed germination ecology of south-western Australian plant communities, Bell *et al.* (1993) found that fire ephemerals, obligate seeders and species cued to flower by fire tended to produce viable, readily germinable seed. They also reported that many clonal, rhizotamous and long-lived post-fire resprouter species often produced mainly inviable seeds and that seedlings were uncommon, even after fire. Pate and Dixon (1982) reported that 80% of tuberous, cormorous and bulbous species bore no evidence of vegetative multiplication so were deemed to depend on seedlings for recruitment. They also observed that, for many species, they rarely observed seedlings, even after fire. This is consistent with our observations where, after four years, the abundance of many of these species was very low compared with unlogged forests, suggesting that recovery to levels comparable with the coupe buffers and the external control are likely to be very slow. Almost all weed species that established after logging were annual grasses or short-lived herbs, so we expect the abundance of these to reduce with time as the cover of native vegetation increases.

Management implications and conclusions

This study has at least two shortcomings. Firstly, it does not provide information about the long-term trajectory of vegetation recovery, and secondly it does not represent the full range of jarrah forest site types. Apart from being important in its own right, we can only speculate about the ecological significance of reduced abundance of native understorey plants and increased abundance and species diversity of weeds measured four years after logging. It is possible that if native plant species abundance does not recover, or recovery is slow, taking perhaps many decades as suggested by Ough and Murphy (1996), then this will have an impact on the energetics of the site in terms of resources available to other organisms and habitat diversity. Excessive soil disturbance may also have adverse impacts on important soil-borne organisms and on processes such as nutrient cycling, hydrology and erosion. While the data reported here represent the situation 4 years post-logging, the biological indications are that recovery of the understorey will be very slow. It could be argued that decisions about the long-term impacts of current silvicultural prescriptions on a range of sites should be known before altering the prescriptions. However, we believe a more prudent approach is to act on the best available knowledge rather than to persist with existing prescriptions for perhaps a further 10 or 20 years, or until the long-term effects are better understood. Because of the relatively recent introduction of contemporary silvicultural prescriptions, further surveys are unlikely to reveal new information about recovery trajectories. Consistent with the precautionary principle and the principle of ecologically sustainable forest management, we believe that logging operations need to be modified to minimise mechanical soil disturbance and damage to the understorey and other non-commercial vegetation. We also suggest that the intensity of silvicultural burns be reduced to minimise soil heating and hence death of understorey species, particularly those with lignotubers and subterranean storage organs.

In an effort to improve the survival of understorey plant populations during logging (clearfelling) operations in wet forests in Victoria, Ough and Murphy (1998) investigated the feasibility of retaining small patches of forest (understorey islands) which can be logged, but from which machinery is excluded. Their trial revealed that the retention of these 'islands' was feasible and that mechanical damage was considerably reduced within the 'islands'. While this may be appropriate for these ecosystems, we recommend modifying logging practices in jarrah forests to minimise soil disturbance, and therefore impact on the understorey, across the entire coupe, not just within 'islands'. This could be achieved by:

1. Reviewing the practice of mechanically disturbing the soil to create a receptive seedbed for commercial tree species;
2. Reviewing the practice of removing some understorey competition by mechanically downing/removing non-commercial tree, lower tree and shrub species;
3. Investigating new systems for accessing, felling and extracting timber that minimise machine traffic on the coupes;
4. Investigating options for utilising machinery with lower ground-bearing pressures;
5. Consider logging only under dry soil conditions;
6. Developing better criteria and codes of practice for acceptable/unacceptable levels of soil disturbance and damage; and
7. On-going monitoring (perhaps five-yearly) to check that the recommended modifications to logging practices are achieving the desired outcome.

Acknowledgements

We thank Dr Ian Abbott, Dr Lachlan McCaw, Keiran McNamara and Matthew Williams (Department of Conservation and Land Management (CALM), Western Australia) for their helpful comments on earlier drafts, Graeme Liddelow (CALM) for his assistance in the field and the Western Australian Herbarium for assistance with plant identification.

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Appendix 1

Plant species collected in 750 × 1 m^2 sample quadrats during a survey of the impacts of logging on jarrah forest understorey vegetation. Formal names follow Chapman *et al*. (2000). Voucher specimens were lodged with the Western Australian Herbarium, Perth. Plant life form (guilds): T = tree, WS = woody shrub, PH = perennial herb, SLH = short-lived herb (<2 years), GR = grass, SD = sedge, AL = alien or weed. Main method of post-fire regeneration where SE = regenerates from seed, WR = resprouts from woody rootstock (lignotuber) or epicormic buds, FBO = resprouts from fleshy below ground organ (roots, bulb, corm, rhizome, tuber), UN = unknown.

Species	Family	Life-form	Post-fire regeneration method
Acacia alata	Mimosaceae	WS	SE
Acacia browniana	Mimosaceae	WS	SE
Acacia extensa	Mimosaceae	WS	SE
Acacia+ mooreana	Mimosaceae	WS	SE
Acacia myrtifolia	Mimosaceae	WS	SE
Acacia pulchella	Mimosaceae	WS	SE
Acacia saligna	Mimosaceae	WS	SE
Acacia urophylla	Mimosaceae	WS	SE
Acaena echinata	Rosaceae	PH-AL	UN
Aira caryophyllea	Poaceae	GR-AL	UN
Amperea ericoides	Euphorbiaceae	PH	WR
Amphipogon amphipogonoides	Poaceae	GR	FBO
Anigozanthos flavidus	Haemodoraceae	PH	FBO
Astroloma drummondii	Epacridaceae	WS	WR
Astroloma pallidum	Epacridaceae	WS	WR
Austrodanthonia caespitosa	Proteaceae	GR	FBO
Austrostipa campylachne	Proteaceae	GR	FBO
Banksia grandis	Proteaceae	WS	WR
Billardiera variifolia	Pittosporaceae	WS	WR
Boronia spathulata	Rutaceae	WS	WR
Bossiaea linophylla	Papilionaceae	WS	SE
Bossiaea ornata	Papilionaceae	WS	WR
Brachyscome iberidifolia	Asteraceae	SLH	SE
Briza minor	Poaceae	GR-AL	SE
Burchardia umbellata	Colchicaceae	PH	FBO
Caesia micrantha	Anthericaceae	PH	UN
Caladenia flava	Orchidaceae	PH	FBO
Caladenia reptans	Orchidaceae	PH	FBO
Cassytha racemosa	Lauraceae	PH	UN
Centaurium erythraea	Gentianaceae	SLH-AL	SE
Chamaescilla corymbosa	Anthericaceae	PH	FBO
Chorizema nanum	Papilionaceae	WS	SE
Clematis pubescens	Ranunculaceae	WS	WR
Comesperma calymega	Polygalaceae	PH	WR
Conostylis aculeata	Haemodoraceae	PH	FBO
Conostylis setigera	Haemodoraceae	PH	FBO
Conyza bonariensis	Asteraceae	SLH-AL	WR
Corymbia calophylla	Myrtaceae	T	WR
Craspedia pleiocephala	Asteraceae	PH	SE
Cyanicula deformis	Orchidaceae	PH	FBO
Cyathochaeta avenacea	Cyperaceae	SD	FBO
Cyrtostylis huegelii	Orchidaceae	PH	FBO
Dampiera linearis	Goodeniaceae	PH	WR
Daucus glochidiatus	Apiaceae	SLH	SE
Daviesia preissii	Papilionaceae	WS	WR
Desmocladus fasciculatus	Restionaceae	PH	FBO
Diuris longifolia	Orchidaceae	PH	FBO
Drosera erythrorhiza	Droseraceae	PH	FBO
Drosera huegelii	Droseraceae	PH	FBO
Drosera menziesii	Droseraceae	PH	FBO
Drosera stolonifera	Droseraceae	PH	FBO
Dryandra bipinnatifida	Proteaceae	WS	WR
Dryandra lindleyana var. *lindleyana*	Proteaceae	WS	WR
Eucalyptus marginata	Myrtaceae	T	WR
Euchiton collinus	Asteraceae	SLH	SE
Gahnia trifida	Cyperaceae	SD	FBO
Galium murale	Rubiaceae	SLH	SE
Gastrolobium bilobum	Papilionaceae	WS	SE
Genus sp.	Lamiaceae	PH	UN
Geranium solanderi	Geraniaceae	PH	SE
Gompholobium marginata	Papilionaceae	WS	SE
Gompholobium tomentosum	Papilionaceae	WS	SE
Hakea amplexicaulis	Proteaceae	WS	WR
Hakea lissocarpha	Proteaceae	WS	WR
Hakea oleifolia	Proteaceae	WS	SE
Hardenbergia comptoniana	Papilionaceae	WS	SE
Hibbertia amplexicaulis	Dilleniaceae	WS	WR
Hibbertia commutata	Dilleniaceae	WS	WR
Hibbertia racemosa	Dilleniaceae	WS	WR
Hovea chorizemifolia	Papilionaceae	WS	WR
Hovea trisperma	Papilionaceae	WS	SE
Hydrocotyle diantha	Apiaceae	SLH	SE
Hydrocotyle sp.	Apiaceae	SLH	SE
Hypochaeris glabra	Asteraceae	PH-AL	UN
Hypolaena exsulca	Restionaceae	PH	UN
Hypoxis occidentalis var. *quadrilob*	Hypoxidaceae	PH	FBO
Isotoma hypocrateriformis	Lobeliaceae	SLH	SE
Isotropis cuneifolia	Papilionaceae	WS	WR
Kennedia coccinea	Papilionaceae	WS	SE
Kennedia prostrata	Papilionaceae	WS	SE
Labichea punctata	Caesalpiniaceae	WS	SE
Lagenophora huegelii	Asteraceae	PH	SE
Lepidosperma leptostachyum	Cyperaceae	SD	FBO
Leptomeria cunninghamii	Santalaceae	WS	SE
Leucopogon australis	Epacridaceae	WS	WR
Leucopogon capitellatus	Epacridaceae	WS	WR
Leucopogon propinquus	Epacridaceae	WS	WR
Leucopogon verticillatus	Epacridaceae	WS	WR
Levenhookia preissii	Stylidiaceae	SLH	SE
Logania serpyllifolia	Loganiaceae	PH	WR
Lomandra caespitosa	Dasypogonaceae	PH	FBO
Lomandra drummondii	Dasypogonaceae	PH	FBO
Lomandra hermaphrodita	Dasypogonaceae	PH	FBO
Lomandra integra	Dasypogonaceae	PH	FBO
Lomandra micrantha	Dasypogonaceae	PH	FBO
Lomandra nigricans	Dasypogonaceae	PH	FBO
Lomandra preissii	Dasypogonaceae	PH	FBO
Lomandra sericea	Dasypogonaceae	PH	FBO
Loxocarya striata	Restionaceae	PH	FBO

Species	Family	Life-form	Post-fire regeneration method	Species	Family	Life-form	Post-fire regeneration method
Lotus suaveolens	Papilionaceae	PH-AL	WR	*Sowerbaea laxiflora*	Anthericaceae	PH	FBO
Luzula meridionalis	Juncaceae	PH	FBO	*Sphaerolobium medium*	Papilionaceae	WS	SE
Macrozamia riedlei	Zamiaceae	WS	WR	*Stackhousia monogyna*	Stackhousiaceae	PH	SE
Millotia tenuifolia	Asteraceae	SLH	SE	*Stylidium adnatum*	Stylidiaceae	PH	SE
Olax benthamii	Olacaceae	WS	UN	*Stylidium amoenum*	Stylidiaceae	PH	SE
Opercularia hispidula	Rubiaceae	PH	WR	*Stylidium brunonianum*	Stylidiaceae	PH	SE
Orchid sp. (petiole leaf)	Orchidaceae	PH	FBO	*Stylidium calcaratum*	Stylidiaceae	PH	SE
Orchid sp. (linear leaf)	Orchidaceae	PH	FBO	*Stylidium carnosum*	Stylidiaceae	PH	SE
Orchid sp. (stripe leaf)	Orchidaceae	PH	FBO	*Stylidium ciliatum*	Stylidiaceae	PH	SE
Orchid sp. (strap leaf)	Orchidaceae	PH	FBO	*Stylidium junceum*	Stylidiaceae	PH	SE
Orchid sp. (ovate leaf)	Orchidaceae	PH	FBO	*Stylidium luteum*	Stylidiaceae	PH	SE
Oxalis corniculata	Oxalidaceae	SLH	SE	*Stylidium rhynchocarpum*	Stylidiaceae	PH	SE
Parentucellia latifolia	Scrophulariaceae	SLH-AL	SE	*Stylidium* sp.	Stylidiaceae	PH	UN
Patersonia babianoides	Iridaceae	PH	FBO	*Tetraria capillaris*	Cyperaceae	SD	FBO
Patersonia occidentalis	Iridaceae	PH	FBO	*Tetraria octandra*	Cyperaceae	SD	FBO
Patersonia umbrosa	Iridaceae	PH	FBO	*Tetrarrhena laevis*	Poaceae	GR	FBO
Pelargonium littorale	Geraniaceae	PH	UN	*Tetratheca affinis*	Tremandraceae	WS	WR
Pentapeltis silvatica	Apiaceae	PH	FBO	*Tetratheca hispidulus*	Tremandraceae	WS	WR
Persoonia longifolia	Proteaceae	WS	WR	*Thelymitra crinita*	Orchidaceae	PH	FBO
Phyllanthus calycinus	Euphorbiaceae	WS	WR	*Thomasia* sp.	Steculiaceae	WS	WR
Pimelea angustifolia	Thymelaeaceae	WS	SE	*Thysanotus* sp.	Anthericaceae	PH	FBO
Pimelea suaveolens	Thymelaeaceae	WS	WR	*Thysanotus multiflorus*	Anthericaceae	PH	FBO
Platysace tenuissima	Apiaceae	PH	SE	*Thysanotus patersonii*	Anthericaceae	PH	FBO
Podolepis canescens	Asteraceae	SLH	SE	*Trachymene pilosa*	Apiaceae	SLH	SE
Pseudognaphalium luteo-album	Asteraceae	SLH-AL	UN	*Tremandra diffusa*	Tremandraceae	WS	UN
				Tricoryne humilis	Asteraceae	PH	FBO
Pteridium esculentum	Dennstaedtiaceae	F	FBO	*Trifolium campestre*	Papilionaceae	SLH-AL	UN
Pterostylis nana	Orchidaceae	PH	FBO	*Trymalium ledifolium*	Rhamnaceae	WS	WR
Pterostylis vittata	Orchidaceae	PH	FBO	*Velleia trinervis*	Goodeniaceae	PH	WR
Ptilotus manglesii	Amaranthaceae	PH	SE	*Veronica calycina*	Scrophulariaceae	PH	FBO
Ranunculus colonorum	Ranunculaceae	PH	FBO	*Vulpia myuros*	Poaceae	GR-AL	SE
Romulea rosea	Iridaceae	PH-AL	FBO	*Waitzia* sp.	Asteraceae	SLH	SE
Scaevola striata	Goodeniaceae	WS	WR	*Wurmbea sinora*	Colchicaceae	PH	FBO
Schoenus grandiflorus	Cyperaceae	SD	FBO	*Xanthorrhoea gracilis*	Xanthorrhoeaceae	WS	WR
Senecio hispidulus	Asteraceae	SLH	SE	*Xanthorrhoea preissii*	Xanthorrhoeaceae	WS	WR
Silene galica	Caryophylaceae	SLH-AL	SE	*Xanthosia atkinsoniana*	Apiaceae	PH	WR
Sollya heterophylla	Pittosporaceae	WS	UN	*Xanthosia candida*	Apiaceae	PH	WR
Sonchus oleraceus	Asteraceae	SLH-AL	UN	*Xanthosia huegelii*	Apiaceae	PH	WR