Skull, Stephen David (1998) ***The ecology of tropical lowland plant communities with particular reference to habitat fragmentation and Melaleuca viridiflora Sol. ex Gaertn. dominated woodlands.*** **PhD thesis, James Cook University**

CHAPTER 4 - THE EFFECTS OF SOIL TYPE AND SOIL MOISTURE ON THE STRUCTURE OF *MELALEUCA VIRIDIFLORA* OPEN WOODLANDS

The lowland soils are highly diverse, and profile wetness, which is usually site dependent, is the major factor influencing soil morphology (Cannon *et al.* 1992).

4.1 INTRODUCTION

4.1.1 General

In the many savanna communities of northern Australia, it has been shown that soil moisture is the major edaphic limitation to productivity (McKeon *et al.* 1990). The authors of this paper also stressed that other variables including temperature and available soil nutrients are also important, particularly when concerned with above ground biomass. Bowman (1986) showed that soil moisture (saturation of the soil profile in the wet season) was the primary determinant of vegetation structure for *Eucalyptus tetradonta* F. Muell. forests of the Northern Territory, with total basal area decreasing on drier sites. Basal area is defined for woody plants as the cross-sectional area at breast height and is usually expressed in square metres per hectare (Shimwell 1971).

Buckman and Brady (1970) discussed several important soil characteristics influenced by soil moisture including:

- soil temperature which is reduced when soil moisture increases, particularly for poorly drained soils;
- soil aeration which is reduced with increasing soil moisture;
- the growth of soil bacteria depending upon whether they are anaerobic, aerobic or facultative bacteria; and
- the rate of nitrification which is retarded by either very high or very low soil moisture.

Coventry and Williams (1984) showed that soil hydrology is related to soil type, with red earths being freely drained, yellow earths waterlogged for short periods and grey earths being the most waterlogged. Grey colouration in the soil profile indicates the presence of organic material, a potential lack of aeration and that waterlogging will occur periodically (Buckman & Brady 1970). No simple link has been shown to exist between soil moisture and soil fertility, and it appears that the water status of soils is more important with respect to plant growth (Dr R. Coventry, pers. comm.). The availability of many soil nutrients is, however, dependent on soil pH (affected by soil moisture) with most nutrients more readily available within a soil pH range of 6-7 (Buckman & Brady 1970).

A literature search recorded only one previous hydrological investigation conducted in *M. viridiflora*-dominated woodlands (Tweddell 1982). In his study, Tweddell (1982) investigated sites close to Townsville (near Sites 1-4 of this study) as well as several in the vicinity of Cardwell (near Sites 16-18 of this study). Using piezometers, it was shown that sites with lower rainfall had water tables well below those of wetter regions towards the end of the dry season. During the wet season, water tables at sites from both low and high rainfall areas were above ground level for some period of time. Numerous authors have recorded that *M. viridiflora* woodlands are usually located in areas of impeded drainage (e.g. Gillison & Walker 1981; Tracey 1982; Stanton & Godwin 1989).

The soil moisture contents and water table depths recorded by Tweddell (1982) are presented in Table 4.1, along with the numbers of the nearest sites examined during this investigation. Soil moisture was unexpectedly high at sites with lower rainfall, and water tables dropped markedly at the end of the dry season seemingly independent of total annual rainfall. In conjunction with the results of the piezometer study, Tweddell (1982) concluded that soil hydrology (including soil moisture) rather than rainfall was important in determining plant community structure and composition. He also found that densities of *M. viridiflora* were highest at sites with duplex soils that usually have poor drainage (Tweddell 1982).

4.1.2 Aim of this investigation

The aim of this section of the thesis was to examine the effects of soil type and soil moisture on the structural attributes of three *Melaleuca viridiflora* open woodland communities.

Table 4.1 **Soil moisture contents (%), water table depths (cm) and approximate annual rainfall (mm) (March-November) for *M. viridiflora* sites investigated by Tweddell (1982).** The nearest sites investigated during this study are also presented. Annual rainfall has been averaged across the relevant sites examined in this study (Chapter 3).

Site (Tweddell)	Nearest sites (This study)	Soil moisture (%) Max. (April)	Min. (Sept.-Oct.)	Water table depth (cm) Max. (April)	Min. (Sept.-Oct.)	Annual rainfall (mm)
1	1, 2, 3 & 4	24	2	5	-20	1400
2	1, 2, 3 & 4	28	10	4	-190	1400
	10, 16	22	18	5	-70	2200
6a/6b	17, 18	28	7	-17	-205	2100

4.2 METHODS

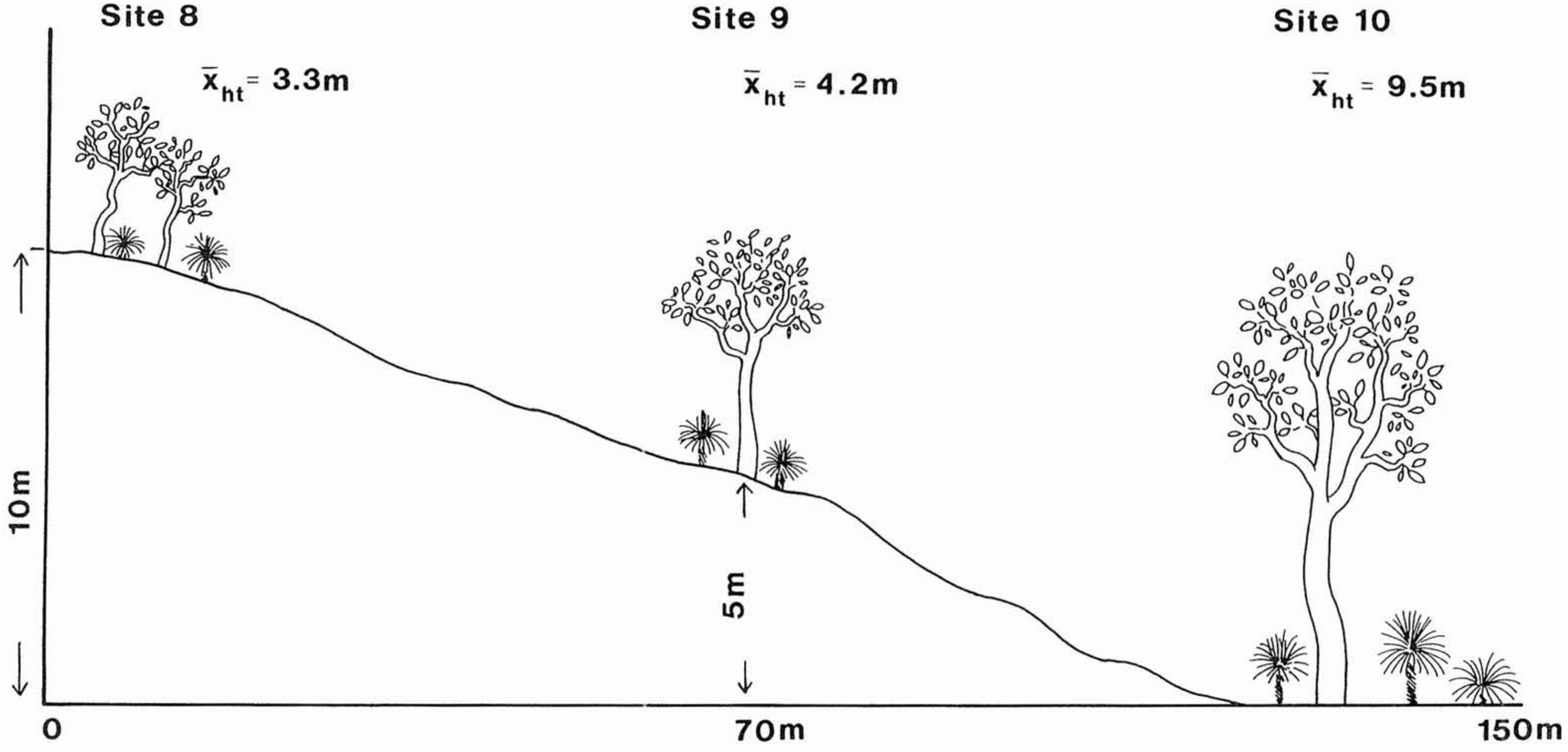
4.2.1 Data collection

Three sites on HINP (Sites 8, 9 and 10) were selected along a transect which was predicted to exhibit changes in soil type, soil moisture, the degree of surface rockiness and, therefore, community structure (see Figure 4.1). This area was also the only field location encountered during this study that satisfied both of the following criteria:

- *M. viridiflora* communities were the only vegetation type present along the transect; and
- no disturbances to hydrological patterns (typical of many mainland communities) were evident.

General community data for these sites are summarised in Chapter 3 (Table 3.4), with the DBH and tree height data (TH) in Appendix C. All sites have had a similar recent fire history.

Figure 4.1 **Transect showing relative tree heights and position of study sites along a perceived environmental gradient.** Data presented (mean *M. viridiflora* height at each site) taken from Table 4.3.



Site 8
$\bar{x}_{ht}$ = 3.3m
Site 9
$\bar{x}_{ht}$ = 4.2m
Site 10
$\bar{x}_{ht}$ = 9.5m
10m
5m
0
70m
150m

A single soil profile was augured to the maximum depth possible at each site as outlined in Chapter 3. At Site 8 the high proportion of rock within the profile made sampling the profile impossible at depths greater than 65 cm. For Site 10 the saturated nature of the soil at 45 cm prevented retention of the auger sample beyond this depth. A profile to a depth of 1 m was completed at Site 9.

Samples for determination of surface soil moisture were collected on six occasions between October 1993 and June 1995. Twenty soil samples (each from the surface; 5 x 5 x 2 cm deep) per site were collected randomly within the vegetation sampling quadrats and returned to the laboratory in pre-weighed airtight jars for analysis (marking the location of each sample ensured no previously disturbed area was sampled on subsequent site visits). Each sample was weighed on a Sartorius electronic balance (± 0.005 g) and then oven-dried for 24 hours at 105°C. The jars were then re-weighed and percentage soil moisture calculated. This gravimetric method of soil moisture determination has proven to be reliable, easy and cheap (Reynolds 1970).

In addition to the data collected previously for these sites, the following community data were recorded (all within the vegetation sampling quadrats):

- the degree of surface rockiness (percentage cover estimated visually from 20 random 50 x 50 cm quadrats);
- the density of *M. viridiflora* seedlings in the understorey recorded from the quadrats used above;
- the heights (mm) of 20 randomly selected *M. viridiflora* seedlings; and
- the heights (cm) of 31 randomly selected *Xanthorrhoea johnsonii* plants.

Both seedling densities and heights were recorded in June 1995, some 45 months after the prescribed fire at Site 5 (see Chapter 5) which also burnt these sites.

4.2.2 Data analysis

A repeated measures ANOVA was used to analyse the soil moisture data collected over the duration of the experiment. A one-way ANOVA with Tukey's HSD post-hoc tests was then used to assess:

- specific time periods of interest;
- community attributes recorded in other sections of the thesis (*M. viridiflora* DBH and height data-Chapter 3); and
- attributes recorded specifically during this investigation (*M. viridiflora* seedling density and *X. johnsonii* heights).

The data set for *M. viridiflora* seedling density was log (x+1) transformed prior to analysis due to the high proportion of zeros recorded (Sokal & Rohlf 1981). As no *M. viridiflora* seedlings were recorded at Site 8, a paired t-test was used to determine if the observed differences in seedling heights at Sites 9 and 10 were statistically significant. A chi-squared heterogeneity test was used to compare the *M. viridiflora* DBH and height size class distributions. Linear regression and Pearson correlation coefficient results (used to examine the relationship between tree height and DBH) have been reproduced from Chapter 3 (Table 3.5). SAS Version 6.0 was used for the repeated measure ANOVA and SPPS Version 6.0 for all other data analysis procedures.

4.3 RESULTS

Each of the soil profiles (Table 4.2) belongs within soil group derived from acid igneous rocks as defined by Cannon *et al.* (1992). Soil types were similar for Sites 8 and 9 (Rungoo) but different at Site 10 (Lugger). The Rungoo profiles were predominantly greyish-brown to light greyish brown in colour, with the upper 0-10 cm of the profile full of massive, sub-angular quartz gravel. All horizons of the profile at Site 9 were much broader than those of Site 8. Further down the Rungoo profiles (>35 cm) soft segregations of manganese and iron were common. Clay content was high even in the upper portions of the B horizon. In contrast, the Lugger soil profile varied from very dark grey near the surface to light grey at depth. Clay was markedly less predominant and finer with no sharp texture contrast recorded for the B horizon. Surface rock was most prevalent at Site 8 and absent from Site 10.

Table 4.2 **Morphological descriptions of the soil profile from auger borings at each of the three study sites.** Soil colour codes from Munsell soil colour charts.

Site 8 (Rungoo)	Site 9 (Rungoo)	Site 10 (Lugger)
A horizon		
0-5 cm: A1-massive surface gravel common (10YR 7/1).	0-10 cm: A1-greyish brown (10YR 5/2) loamy sand, massive surface gravel occasional.	0-10 cm: A1-very dark grey (10YR 3/1) coarse sandy loam.
5-15 cm: A2-clayey sand (10YR 7/2).	10-35 cm: A2-white to light brownish grey (2.5YR6/2), coarse sand, some clay content evident near max. depth.	10-35 cm: A2-dark grey (10YR 4/1) coarse sandy clay loam.
B Horizon		
15-35 cm: B2-medium heavy clay (2.5YR 6/2) with yellow mottles predominant.	35-70 cm: B21-light brownish grey medium heavy clay. Yellow mottles predominant, manganese nodules present.	35-? cm: B2-light grey (10YR 7/3) coarse sandy clay loam, coarse gravel infrequent.
35-65 cm: 2Bb-sandy medium clay (7.5YR 6/8) with manganese and iron rich soft segregations common.	70-100 cm: B22-as above but sandy light clay.	The profile was difficult to auger past 35 cm due to the saturated nature of the soil.
65-? cm: Difficult to auger past this depth due to high rock content.	1 m was the maximum auger depth possible.	

A summary of the data collected at the three study sites is presented in Table 4.3. Mean soil moisture (for the entire sampling period), *M. viridiflora* DBH, height, seedling density, and seedling height and *X. johnsonii* height were all lowest at Site 8 and highest at Site 10, with Site 9 intermediate between the two (Table 4.3). The density of *X. johnsonii* was greatest at Site 9, markedly less at Site 8 and almost absent from Site 10.

Table 4.3 **Summary of the community data collected at each of the three sites.** Where appropriate standard errors are given in parentheses. Soil types follow those of Cannon *et al.* (1992).

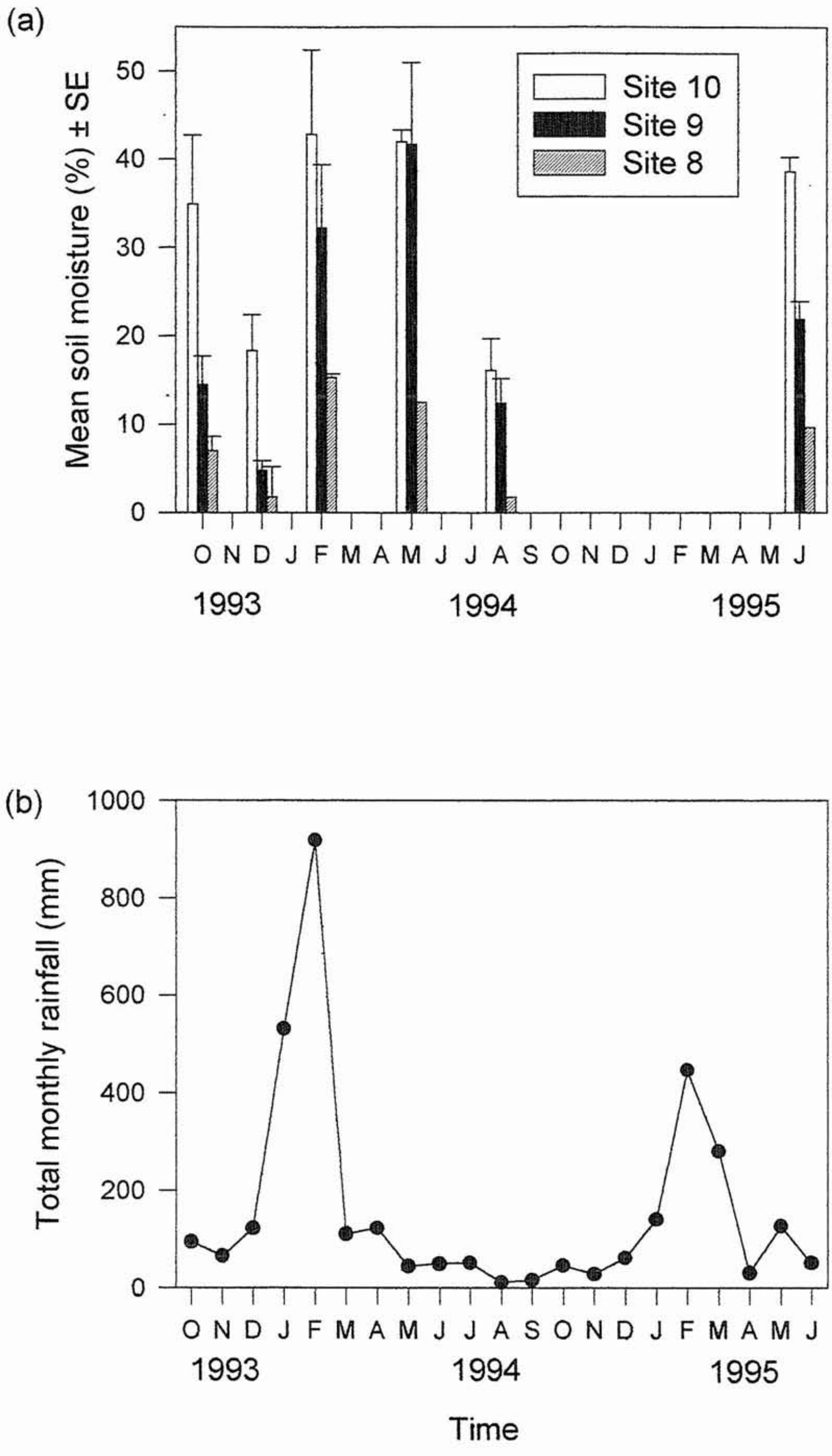
Parameter	Site 8	Site 9	Site 10
Soil type	Rungoo (duplex soil)	Rungoo (duplex soil)	Lugger (gradational soil)
Percentage surface rock	40	15	0
Mean % soil moisture (Pooled for entire sampling period)	8.0 (0.7)	21.2 (1.9)	32.0 (1.1)
Melaleuca viridiflora			
Mean DBH (cm)	4.3 (0.3)	5.9 (0.3)	13.2 (0.6)
Mean height (m)	3.3 (0.2)	4.2 (0.2)	9.5 (0.6)
Basal area (m^{-2} ha^{-1})	16.83	9.29	4.71
Tree/sprout density (500 m^{-2})	53/154	142/154	55/94
Mean seedling density (m^{-2})	0	0.8 (0.4)	1.9 (0.6)
Mean seedling height (cm)	0	6.6 (0.6)	16.9 (1.3)
Xanthorrhoea johnsonii			
Density (500 m^{-2})	175	725	30
Mean height (cm)	0	15.8 (2.8)	34.8 (4.3)

Surface soil moisture data for each site and sampling time are presented in Figure 4.2. Soil moisture was always lowest at Site 8 and highest at Site 10, with records for Site 9

Figure 4.2 **Mean percentage soil moisture (%) and rainfall (mm) at the three study sites over a 20 month period.**

(a) Soil moisture (%).

(b) Rainfall (mm).


(a)
Mean soil moisture (%) ± SE
0
10
20
30
40
50
Site 10
Site 9
Site 8
O N D J F M A M J J A S O N D J F M A M J
1993
1994
1995
(b)
Total monthly rainfall (mm)
0
200
400
600
800
1000
O N D J F M A M J J A S O N D J F M A M J
1993
1994
1995
Time

intermediate between the two. The lowest soil moistures were recorded following extended dry periods in the region (determined from Bureau of Meteorology data supplied for Chapter 3). Highest soil moistures were recorded during and just after a significant rainfall event (February 1994, May 1994 and June 1995 samples). Maximum soil moistures were higher than those documented by Tweddell (1982), yet minimums fell within a similar range.

The repeated measures ANOVA results are summarised in Table 4.4, and indicate that the ranking of sites (8 driest, 9 intermediate and 10 wettest) is always maintained (all results were statistically significant with $p<0.0001$). Temporal changes occur in the magnitude of the difference as demonstrated by the F ratios, which are calculations of the between group variation scaled by the within group variation (McArdle 1994). An examination of sphericity (a test of whether the error correlation matrix is significantly different from an identity matrix) for the repeated measures ANOVA indicated that the assumptions of the ANOVA had not been violated.

Results from the Tukey's HSD post-hoc tests show that for two sampling occasions (May 94 and August 94), despite there being an overall significant result for all sites collectively, Site 9 is not significantly different from Site 10 (Tables D1 and D2, Appendix D).

Significant differences between sites were recorded for *M. viridiflora* DBH ($p<0.0001$), *M. viridiflora* heights ($p<0.0001$) and *X. johnsonii* heights ($p<0.001$), with Site 8 and 9 both significantly different (lower) from Site 10 (Tables D3-D5, Appendix D). Densities of *M. viridiflora* trees were nearly three times higher at Site 9 compared with Sites 8 and 10, whilst sprout density was highest at sites with duplex soils (Sites 8 and 9).

The size class data, regression calculations and correlation coefficients for tree height and DBH data are summarised in Table 4.5. The chi-squared comparison of both *M. viridiflora* DBH and height size class data also recorded significant differences between all sites (Table D6, Appendix D). For the tree height data, Site 10 (most different from Sites 8 and 9) had higher numbers of trees in the larger size classes (mode in TH 5,

$11<x\leq14$ m), whilst Sites 8 and 9 had larger numbers of trees in the three smallest size classes (modes both in TH 2, $2<x\leq5$ m).

Table 4.4 **Results of repeated measures ANOVA for soil moisture.** All results were significant with $p<0.0001$. F ratios for effects are the Wilks' Lambda statistic.

Test	df	F ratio
October 1993	2	361.58
December 1993	2	271.49
February 1994	2	71.83
May 1994	2	81.09
August 1994	2	24.20
June 1995	2	81.09
Time effect	5	254.14
Time*site effect	10	20.36
Between subject effects	2	661.40
Within subject effects (time)	5	146.95
Within subject effects (time*site)	10	16.88

The structural similarity of Sites 8 and 9 (both on duplex soils) was also confirmed by the DBH data. Site 10 had a large proportion of trees (71%) in three of the larger size classes (mode in DBH 4, $10<x\leq15$ cm), with Sites 8 and 9 having the majority of trees (97 % and 91% respectively) in the three smallest size classes (modes both in DBH 2, $x\leq2$ cm). All possible structural size class comparisons were above the critical χ^2 value for the appropriate number of degrees of freedom, and p values were all less than or equal to 0.01 (Table D6, Appendix D).

Table 4.5 **Size class structural data, regression and correlation coefficient (height = a + b DBH) for each site from Chapter 3.** The modal frequency is highlighted in bold. A "-" indicates no individuals in a category. No data for any category above DBH 7 were recorded (Appendix C).

	DBH structural size class							Regression			Correlation coefficient
Site	1	2	3	4	5	6	7	r^2	b	a	r
				DBH (cm)							
8	4	**68**	29	2	1	-	-	0.75	0.63	0.95	0.87
9	17	**62**	50	10	2	1	-	0.76	0.55	1.40	0.87
10	15	2	10	**25**	15	2	1	0.61	0.53	4.46	0.78
				Height (m)							
8	12	**72**	15	4	1	-	-				
9	20	**78**	4	7	3	-	-				
10	5	13	7	13	**22**	10	-				

The correlation coefficients for Sites 8 and 9 were identical. The slopes of the regression lines are, however, quite different with Site 8 having the steepest positive slope. The regression line slope was slightly lower for Site 10 compared with Site 9. Site 10 had a markedly lower correlation coefficient than both the other sites indicating that the relationship between tree height and DBH is less tightly correlated.

No *M. viridiflora* seedlings were recorded from Site 8. The t-test to compare the heights of the seedlings at Sites 9 and 10 recorded a significant difference (t=8.28, p<0.001). The one-way ANOVA for the log (x+1) transformed seedling density data also recorded a significant result (p<0.001), with both Site 8 and 9 recording means statistically different (lower) from that of Site 10 (Table D7, Appendix D).

Basal area was significantly different at each site ($p<0.0001$) with the lowest basal area recorded at Site 8 and the highest at Site 10 (Table 4.3). The results from the Tukey's HSD post-hoc test also supported the result that Site 10 was the most different from Site 8 (Table D8, Appendix D).

4.4 DISCUSSION

The Rungoo series (Sites 8 and 9) of soil profiles characteristically have a sharp texture break from the A to the B horizon, with the B horizon often exhibiting a high clay content (duplex soil). They are also highly infertile (Cannon *et al.* 1992), a feature they have in common with many tropical woodland soils where phosphorus (along with other critical elements) may limit plant growth (Salisbury & Ross 1985). The Rungoo profile recorded at Site 8 had even shallower horizons, and higher clay and rock contents (Table 4.2) than those described by Cannon *et al.* (1992) which supported stunted to tall *Melaleuca* forest.

This type of profile is therefore expected to restrict plant growth, particularly when soil moisture levels are low for most of the year. As a result, nutrient availability would also be predicted to be low, especially when the profile is relatively well drained in topographic terms. Additionally, it has been observed that roots tend not to penetrate the heavy clay sections of the profile, thus restricting the available sub-surface space for root growth (Tweddell 1982; Dr R. Coventry, pers. comm.). Site 9 had a similar profile to that documented by Cannon *et al.* (1992), and it appeared that the broader horizons, higher surface soil moisture contents and lower rock proportions provided improved conditions for *M. viridiflora* growth. This site also had higher seedling densities than Site 8, suggesting that there may be better conditions for seed germination and seedling establishment of this and other (*X. johnsonii* was most prolific here) woodland species. Caution must be exercised in interpreting the results, however, as seedling density may vary over time in any given soil type.

The Lugger soil series (recorded at Site 10), however, typically exhibit a gradational texture profile (Cannon *et al.* 1992). As a result, soils belonging to this series are usually better drained (where topography allows) and more fertile, although both Lugger and Rungoo soils have been documented as supporting tall *Melaleuca* forests (Cannon *et al.* 1992). Cannon *et al.* (1992) also found that the Lugger profiles they examined had higher levels of available phosphorus, nitrogen and increased exchange properties when compared with the Rungoo series. These characteristics were particularly evident near the surface of the profile where fine root growth of *Melaleuca* species can be extensive (Tweddell 1982).

The grey colour of the Lugger profile recorded at Site 10 indicates that waterlogging will occur occasionally, that there is organic matter present in the profile and that aeration may be low at certain times (Buckman & Brady 1970). The presence of other typical swamp species at Site 10 (*Lophostemon suaveolens* and *Pandanus tectorius* Parkinson, Appendix B), and field observations of standing surface water at this site following substantial wet season rains, further support this hypothesis. Although waterlogging reduces aeration and the rate of certain soil chemical processes (e.g. nitrification), the presence of higher levels of organic matter coupled with high water availability (without waterlogging) for most of the year may compensate for this seasonal occurrence. It is perhaps not surprising then that given the nature of the profiles recorded, Site 10 (with the highest soil moisture and Lugger soil type) had taller trees with larger DBH's (Table 4.3). The Grasstrees (*X. johnsonii*) at Site 10 were also taller than those of other sites, although fewer were recorded. This reduction in Grasstree numbers may indicate that conditions favouring optimal germination and establishment (e.g. repeatedly high soil moisture) of this species are not usually present at this site.

Despite the large difference in sizes of the trees at the three sites, the trees all appear to be at least similar in terms of their respective ages. The Tea-tree orchid (*Dendrobium canaliculatum*), whose approximate age can be determined by counting the number of pseudobulbs present (Mr T. Bartareau, pers. comm.), was recorded at all three sites At each site, individual orchids growing on upper tree branches were estimated to be well over 20 years old. This indicates the likelihood that even the very small tress at Sites 8 and 9 are in fact mature, and that tree size (small or large) is not simply a measure of age, but also a reflection of the soil type and conditions in which they grow.

Interestingly, despite the significant structural and soil differences between Sites 8 and 10, they were grouped by the classification analysis of species presence/absence data (Chapter 3).

Site 10 also had the highest level of *M. viridiflora* recruitment (seedling density) in the understorey. From a study of the effects of soil moisture (at constant temperature) on the germination of *M. viridiflora* (see Chapter 6), it has been documented that relatively high levels of germination are maintained in soils containing between 10% and 40% water. Only at zero soil moisture levels was germination significantly affected. Soil moistures close to zero were recorded twice for Site 8 (December 1993 and August 1994) and fell below 10% once (December 1993) at Site 9 (Figure 4.2).

Tweddell (1982) recorded that *M. viridiflora* tree density was highest on sites with duplex soils. This pattern was supported by the findings of this study in terms of the number of sprouts (highest at both Sites 8 and 9) and trees (although only at Site 9) present at particular sites. The range of basal areas documented at the three sites (4.71-16.83 m^{-2} ha^{-1}) is similar to those recorded for *Eucalyptus tetradonta* forests in the Northern Territory located along a comparable, pronounced soil moisture gradient (Bowman 1986). The figures remain well below those documented for tall eucalypt forests (e.g. 26.1 m^2 ha^{-1} for the Karri forests of south-western Australia, O'Connell 1987). The finding that basal area was highest on wetter sites also concurs with that of Bowman (1986).

The lower basal areas of drier sites may also be a result of more regular burning. Although the recent fire history at all three sites is believed to be similar (see Chapter 3), this may not have always been the case. In some instances fire may have burnt up to the margin of the damper site before being extinguished. The presence of high densities of *X. johnsonii* at the intermediate site (Site 9), however, indicate that once within this community, fire would usually be intense and unlikely to be extinguished even in an area with increased surface soil moisture. The potential for high intensity fires where Grasstrees are a significant component of the understorey results from the highly inflammable nature of their foliage (Vines 1968).

Soil moisture and soil type thus appear to have a significant effect on the structure of *M. viridiflora*-dominated woodlands. It is therefore important that any study attempting to assess the structure of these woodlands has an understanding of these variables. In Chapter 3, however, soil type only partially explained the observed classification and ordination groupings of sites. Other soil profile characteristics such as fertility and nutrient availability are also likely to be important with respect to the control of plant growth, although these were not investigated during this study. Fire, the major natural disturbance occurring within these communities is also believed to play a significant role in the regulation of community structure, an hypothesis which is investigated in more detail in Chapter 5.

CHAPTER 5 - THE EFFECT OF FIRE ON THE STRUCTURE AND FUEL LOADS OF *MELALEUCA VIRIDIFLORA* OPEN WOODLANDS

The capture of fire by the genus Homo *changed forever*
the natural and cultural history of the Earth (Pyne 1991).

5.1 INTRODUCTION

5.1.1 Fire and ecosystem disturbance

General effects of fire

Pickett *et al.* (1989) defined disturbance as "a change in ecosystem structure caused by factors external to the hierarchical level of the system of interest". Disturbance and the function of terrestrial ecosystems have been comprehensively reviewed by Pickett and White (1985) and Attiwill (1994). The latter considers natural disturbance by fire to be the dominant force determining the evolution and development of terrestrial ecosystems, particularly in countries such as Australia (Attiwill 1994). When discussing the ecological effects of fire it is important to understand the concept of a fire regime, which was first defined by Gill (1975) as being comprised of three components; fire intensity, fire season and fire frequency. All three factors are considered critical from an ecological viewpoint as a succession of fires, rather than a single fire event, has the potential to exert a strong influence on the success of an individual, population or species (Gill 1981).

It is accepted that fire has always been part of the Australian landscape, with many native plant communities being prone to recurrent fires (Keith 1996). Lightning-ignited fires began to shape our flora and fauna communities long before human occupation in Australia (Singh *et al.* 1981), and lightning continues to be a major source of ignition in many areas (Martin 1996). Other aspects of the history of fire on the Australian continent are, however, far less certain, and continue to be widely debated within the literature.

The date of the Aborigines' arrival in this country remains uncertain, with estimates ranging from anywhere between 40,000 and 120,000 ago (Kershaw 1985). Following their arrival in tropical Queensland up to (and possibly more than) 60,000 years ago (Crowley 1994), it is generally agreed that fires became a more predominant feature of the Australian landscape. Burning continues to play an important role in Aboriginal culture. Its uses include encouraging new grass growth to attract animals for hunting, increasing access to other food sources, and the creation and maintenance of paths for travelling (Jones 1969).

The scale and effects of Aboriginal fires are, however, the subject of much debate. Some researchers have argued that extensive areas of northern Australia were burnt seasonally by Aboriginal people, promoting the expansion of dry sclerophyll forests and grasslands, and producing a relatively stable mosaic of habitats (Nicholson 1981; Hallam 1985; Press 1987). This argument is supported by some interpretations of pollen records determined from sediment cores (sclerophyllous species replacing fire sensitive species at approximately 60,000 years, Kershaw 1985) and the early records of Australian explorers.

Other researchers have highlighted the inconsistencies of the pollen record and the simplistic nature of the subsequent vegetation-fire-man interaction hypothesis (Horton 1982; Pyne 1991). More recently it has been suggested that other factors such as the megafaunal extinction (occurring between 24,000 and 60,000 years ago) caused massive changes to fuel loads across the country and consequently the resultant fire regime (Flannery 1994). The generally accepted notion that fires occurred across large sections of the landscape are being challenged by current research with Aboriginal people within the WTBR (Ms R. Hill, pers. comm.).

The view that large scale fires were not in the interest of the Aboriginal people is not new, as it is believed they would have caused massive loss of resources (Horton 1982) and warring with neighbouring tribes (Ms R. Hill, pers. comm.). Whatever the history and cause of vegetation change over this time period, it has been validated by local Aboriginal people that their use of fire certainly included the burning of *M. viridiflora* woodlands (Mr R. Butler, pers. comm.). These burning practices were permanently interrupted following European settlement, with large tracts of land subsequently being left unburnt (Florence 1994). This has resulted in massive and continuing ecosystem changes country-wide, nowhere less so than the woodlands of north-eastern Queensland (Lewis 1989; Stanton 1992; Innis 1994).

The protracted debate on the history of fires in Australia, while useful in setting a contemporary framework for fire management in terrestrial landscapes, tells us little about the actual direct effects this disturbance has on plant species or communities. Recent publications have stressed the need for research to concentrate on increasing our understanding of these effects (Whelan 1995; Williams & Gill 1995). Nowhere is this need more evident than in the tropics of north-eastern Queensland, where published scientific information on the effects of fire is almost non-existent.

The ecological effects of fire on Australian plant communities have, however, been reviewed several times (e.g. Gill 1975, 1981, 1996; Gill *et al.* 1991; Whelan 1995; Keith 1996), as has the management of fire in conservation reserves across Australia (Christensen & Maisey 1987; Press 1987; Gill & Nichols 1989; Lewis 1989; Wardell-Johnson *et al.* 1989; Preece 1990; Stanton 1989, 1992; Williams & Gill 1995; Conroy 1996). The Bibliography of Fire Ecology in Australia (Gill *et al.* 1994) provides an additional 2000 references related to the effects of fire and its management in Australian terrestrial ecosystems. Temperate ecosystems are predominantly the focus of the reviews and research conducted within Australia, with tropical eucalypt savanna communities in the Northern Territory a notable exception. The difficulties in comparing research results, even between tropical regions, has already been discussed (Chapter 3), and only changes to vegetation structure and understorey composition (particularly *X. johnsonii*) will be considered further here.

Responses of key M. viridiflora *woodland species to fire*

Gill (1981) classified the way in which species respond to and survive intense fire, placing them in either an obligate seeder (non-sprouter) or sprouter category. The responses to fire of some ten percent of the vascular flora of Australia have been documented on a national register (Gill & Bradstock 1992). Unfortunately, the register currently contains no information regarding *M. viridiflora* (Dr M. Gill, pers. comm.), and like much fire-related research conducted within Australia, its records are predominantly from temperate systems. The potential responses of plants to fire are numerous and have been discussed fully by other authors (Gill 1981; Whelan 1995). One of the only references to fire and *M. viridiflora* woodlands comes from Stocker and Mott (1981), who described these communities as being adapted to regular fire.

Like eucalypts, the seeds of *M. viridiflora* are protected within a woody capsule (sometimes a characteristic indicative of obligate seeders), with this species capable of producing cohorts of seedlings following fire (pers. obs.). *Melaleuca viridiflora* is also capable of sprouting vigorously following fire (Tweddell 1982). This sprouting ability predominantly stems from a lignotuber, although epicormic buds are also utilised (pers. obs.). *Melaleuca viridiflora* thus has several responses to fire, and can therefore be classified as a facultative resprouter (Keeley 1986).

Several papers have detailed the effects of fire on southern species of the genus *Xanthorrhoea* and their adaptations to fire. These include:

- leaves only being killed by fire after which the pre-fire shoot apex continues to grow. This compares with some dicots whose subterranean buds can be released from inhibition following fire (Gill 1981); and
- the apical meristem lying protected up to 12 cm below the soil surface, numerous basal leaf meristems which are protected by tightly packed leaf bases and increases in flowering intensity following fire (Gill & Ingwersen 1976).

The response of a northern species of the genus (*X. johnsonii*) was studied as it was often predominant in the understorey of the *M. viridiflora* communities examined (Chapter 3). The growth and flowering history of this species has been discussed by Bulow-Olsen *et al.* (1982), who documented its slow growing nature and increases in understorey density following fire in a eucalypt-dominated community.

The effects of fire and plant populations

The tolerance of different plant species to fire varies greatly, reflecting the evolutionary strategies developed by plants to survive a variety of fire regimes (Nambiar 1985). Plant growth, reproduction and survival are directly affected by fire (Bond & van Wilgen 1996). Fire tolerance results from fire resistance and an ability to recover following fire (Gill 1975). Plant resistance to burning is determined by the fire regime, plant developmental stage, and the presence or absence of adaptive traits (Braithwaite & Estbergs 1985).

Ecological modellers have attempted to predict the fate of both obligate seeders and sprouters subject to recurrent fires, but the task is complex and fraught with potential problems (Williams & Gill 1995). Noble and Slatyer (1980) provided a predictive tool for analysing changes in plant populations following fire, but their vital attribute model only provided resolution at the species presence/absence level. A subsequent model allowed more detailed assessments of stand structure to be made, as it took into account not only life history attributes, but also competition (responses of plants to their immediate environment) and responses to natural disturbances (Moore & Noble 1990). Frequent fires over a medium to long time period (100 years) were predicted to have significant effects on temperate woodland structure. The study was also modified to examine similar effects on tropical eucalypt communities in Kakadu National Park, but unfortunately these data remain unpublished (Moore & Noble 1990).

Tolhurst (1996), and Bond and van Wilgen (1996) concluded that recurrent fires had more pronounced effects on stand structure than single fires, with the possibility of trees being eliminated should fires become too frequent. Fire frequency was also found to be the most important component of the early model developed by Noble and Slatyer (1980). Further, even prior to the 1970's, Jackson (1968) showed that frequent burning produced understocked forests, and in extreme cases caused the elimination of trees.

Bradstock (1990) used demographic information to predict trends in the population dynamics of a serotinous, resprouting (facultative) tree under a range of fire regimes. His study emphasised the importance of fire intervals of sufficient length to allow for the survival of seedling regeneration from seed stored in the soil (seed bank), and replenishment of the soil seed bank. Importantly, he predicted the likely decline of facultative resprouters under fire regimes with high fire frequencies, even though facultative resprouters were more likely to persist than obligate seeders. Fox and Fox (1986a) also concluded that areas burnt too frequently could lose obligate seedling regenerators, whilst areas left unburnt for extended lengths of time may lose obligate resprouters.

Fire and the lack of it can both be responsible for the localised extinction of plant populations (Keith 1996). In his review of fire-driven extinction of plant populations, Keith (1996) describes seven major mechanisms of decline/extinction. Relevant to this study (presented in Table 5.1) are those associated with high frequency, low intensity fire regimes as these have been documented as the prevalent types of fire in tropical northern Australia (Walker 1981; Ridpath 1985). This is supported by Lonsdale and Braithwaite (1991) who state that "compared with temperate regions, fires in the tropics are usually more frequent and of lower intensity, as the vegetation is generally less flammable and grass fuel rarely accumulates to high levels".

The effects of single fires on plant population structure have been more widely studied, with stand structure often used to assess regeneration processes in plant communities (e.g. Fensham & Bowman 1992). In temperate regions the effects of fire on eucalypt communities are well documented, with fire frequency being critical in determining demographic structure (Bowman & Kirkpatrick 1984; Bowman 1986). In a study of temperate eucalypt forests, Shea *et al.* (1981) concluded that fire can drastically alter the structure and composition of these plant communities. Attiwill (1994) noted that fire could relieve stand stagnation, i.e. restore structural diversity through the successful recruitment of seedlings and saplings.

Table 5.1 **Fire-driven mechanisms of plant population decline and extinction potentially most relevant to *M. viridiflora* communities in this study (after Keith 1996).**

Life-cycle process	Fire regime characteristic	Mechanism of decline/extinction	Life-history types affected
1. Death of standing plant and seeds.	High frequency fires.	Depletion of standing plants through depletion of bud banks, starch reserves or structural weakening.	Resprouters and passive fire tolerators.
2. Failure of seed release and/or germination.	Low intensity fires.	Low rate of recruitment due to release of few seeds.	Species (especially trees) with serotinous seed banks and heat-dependent seed release.
3. Failure of seedling establishment.	High frequency fires.	Physical change to habitat rendering it less amenable to seedling survival.	Species on highly organic or erodable substrates.
4. Interruption of maturation and developmental growth.	(i) High frequency fires.	Fire-induced death of pre-reproductive juvenile plants.	Obligate seeders affected more rapidly than resprouters.
	(ii) High frequency fires.	Fire-induced death of pre-resistant juvenile plants.	Resprouters and passive fire tolerators.

Sapling recruitment is also controlled by other important ecological processes. In temperate regions, the control of sapling recruitment by the suppressive influence of the overwood (or canopy) on lignotuberous growth appears common (Bowman 1986). Intraspecific competition between juveniles has also been demonstrated to suppress eucalypt sapling growth in tropical eucalypt savannas (Fensham & Bowman 1992). The ecological effects of fire on both the floristics and structure of eucalypt communities in tropical Australia have, however, been the subject of intense and sometimes acrimonious debate (Duff & Braithwaite 1989). This has centred on the probability of rainforest species invading eucalypt savannas in the absence of fire, the interpretation of the effects of single and repeated fires and differences of opinion regarding the statistical validity of various components of different research projects (discussed further below).

Several authors have noted the lack of sapling and pole-sized trees in eucalypt savanna communities (Werner 1986; Bowman & Wilson 1988; Fensham & Bowman 1992). Bowman (1986) and others (Wilson & Bowman 1987; Bowman *et al.* 1988) concluded that although the removal of overwood (e.g. treefall) and site quality (especially environmental stress) were both critical for sapling recruitment, the process of recruitment could be independent of fire regime. This conclusion has been emphatically disputed by other scientists (Lonsdale & Braithwaite 1991), who investigated the short-term response of a eucalypt community to an unusually intense fire and reported significant structural alterations. Braithwaite and Estbergs (1985) decided that the poor representation of saplings in the midstorey and canopy was a result of frequent, poorly-timed fires. Fensham (1990) also reported that tropical eucalypt communities exhibited structural changes in the absence of fire, and that the total number of saplings increased from annually-fired sites through to a maximum at fire-free sites.

Bowman and Panton (1995) compared previously studied sites that had been protected from fire for 20 years with those subject to ambient fire regimes. Their results support the initial conclusion of Braithwaite and Estbergs (1985) that fire is responsible for the absence of a well-developed sapling layer in eucalypt savannas. However, other factors including the influence of overwood were still important determinants of community structure.

Thus, even in tropical communities that have been the subject of large, long-term research programs (the design problems of which have been acknowledged by both sides of the debate), discrepancies exist in the interpretation of the precise nature of the effects of fire on community structure. Bowman (1992) notes that little information exists regarding the effects of fire on tropical floodplain communities (including *Melaleuca* woodlands), although this is the focus of research in progress within Kakadu National Park (Dr J. Russell-Smith, pers. comm.). In his recent review of the responses of plant communities to fire, Whelan (1995) noted the need (amongst others) for the following important fire research:

- a comprehensive review of plant succession as it relates to fire ecology examining, in particular, change with respect to different fire histories; and
- documentation of the changes in community parameters with replication of fires.

This thesis therefore sought to address the latter with respect to the structure of *M. viridiflora* woodlands, and primarily refers to the detailed review of Keith (1996) regarding fire and its effects on plant populations. Additionally (for reasons outlined in the subsequent sections), it aimed to examine the temperatures reached during a prescribed fire in a *M. viridiflora* woodland, and the effects of such fires on fine fuel loads.

5.1.2 Fuel load accumulation

The fuel load present within a plant community (fuel availability) refers to the amount of fuel that will burn in a fire, and can be comprised of leaf material, flowers, fruit, twigs, branches and logs (Walker 1981). For the purposes of this study, fuel load refers only to the fuel represented in the fuel bed (ground litter) and small shrub (to 1.5 m) layer. In an international review of fuel load dynamics in terrestrial ecosystems, Facelli and Pickett (1991) reviewed the effects of litter on plant community structure and dynamics. These include the effects of fuel on germination and establishment, vegetation structure, species richness, species composition and productivity. Temporal variation in litter quantities and patterns of litter accumulation (considered in more detail below) were also discussed.

Studies of fuel dynamics within Australia have been prevalent in the literature for at least the past 35 years. Eighty research papers relevant to fuel dynamics were examined during this study, and a summary of where this research has been conducted is presented below in Table 5.2. Little research has been conducted in tropical Queensland on the fuel dynamics of its numerous vegetation types, with three of the five papers describing work in rainforests. The fuel dynamics of the tropical lowland ecosystems of north-eastern

Australia therefore remain poorly documented.

Table 5.2 Area of study for 80 fuel dynamics research papers published in Australia.

Qld (subtropics)	Qld (tropics)	NSW	WA	NT	VIC	TAS	SA	ACT
7	5	10	21	10	9	4	2	12

Fuel characteristics (including fuel loads) correspond with vegetation types that have certain fire regimes (Bowman & Wilson 1988). Walker (1981) recorded that ten out of twelve Australian studies showed fuel peaks occurring in summer. Fuel loads have also been correlated with canopy projective foliage cover (Walker 1981; Marsden-Smedley & Catchpole 1995). Perhaps the most comprehensive compilation of the causes of variation in fuel loads within vegetation types is provided by Fensham (1992), who lists the following causes: seasonal conditions; tree cover; vegetation patchiness; rainfall (with fuel loads higher in wetter areas or seasons); soil fertility; grazing; insect herbivory; slope; and patchiness of previous burning in a given area. It is also worthwhile noting that at the time of ignition, fuel availability is subject to variations in fire intensity, fuel moisture content, flammability of the fuel, size and the degree of fuel compaction (Walker 1981).

An abundance of techniques have been developed to describe fuel types and measure fuel loads. These have, amongst others, included:

- visual methods (Luke & McArthur 1978; Schneider & Bell 1985);
- photographic methods to characterise fuel types (Fahnestock & Key 1971; Wigston 1993);
- characterisation of understorey fuels based on vegetation structure and species composition (Peet 1971);
- estimation of fuel loads via remote sensing (Williamson 1991);
- semi-quantitative estimates using slides and disks (Sneeuwjagt 1973; Schneider & Bell 1985); and
- quantitative harvest methods (numerous studies including this investigation).

Unfortunately, although harvest sampling is now a widely used technique for examining fuel accumulation patterns, no national standard has been adopted by fire researchers in Australia. High levels of variability are evident within the literature with respect to the size of the sampling quadrat, the size class of fuel sampled and the number of samples collected (Table 5.3). This variability remains despite repeated calls from fire modellers for national standards (particularly for the maximum particle size sampled) to be implemented (e.g. McCaw 1991). Such standards would allow for more direct comparisons to be made between fuel dynamics studies, both here in Australia and overseas. Standardisation may, however, be difficult to implement in practical terms. For example, different sized quadrats may be suited to particular vegetation types. Furthermore, a compromise between data accuracy (increased with larger quadrats and higher numbers of samples) and resource availability must usually be reached.

The relationship between litterfall and fuel accumulation was first described by Olson (1963). His exponential model, based on data obtained in a variety of evergreen forests around the world, had the following form:

$$W_t=W_{SS}(1-e^{-kt}),$$

where: W_t=weight at time t (time since fire measured in years); W_{SS}=fuel load under steady-state conditions and k=a decomposition constant. Olson's model has been adapted and repeatedly validated by numerous researchers in Australia including Raison *et al.* (1983), Burrows (1985), Burrows and McCaw (1990) and Fensham (1992). The model has recently been refined to describe seasonal and annual variations in decomposition and accession by Mercer *et al.* (1995). The general equation proposed by Olson (1963) still provides a relatively accurate fit to the hypothetical variations proposed by Mercer *et al.* (1995). Other models of fuel accumulation have also been proposed For example, Hutson and Veitch (1985) put forward a polynomial equation and van Loon (1977) adopted a linear function to define fuel accumulation. These models, however, have not been widely adopted, and Olson's model remains the most conceptually coherent (Fensham 1992). Exponential fuel accumulation curves have now been documented for a wide range of plant communities throughout Australia (for example Raison *et al.* 1983).

Table 5.3 **Some sampling strategies within the literature for quantitative (harvest) fuel load estimations.** An asterisk (*) indicates the strategy adopted in this study.

Quadrat size utilised (m^2)	Maximum fuel particle size sampled (mm)	No. of samples collected
0.04-Burrows & McCaw (1990)	6-Burrows & McCaw (1990); Fensham (1992)	5-Finlayson *et al.* (1993); Williams *et al.* (1995)
0.0625-van Etten (1995)	15-Bowman & Wilson (1988)	10*-Burrows & McCaw (1990)
0.1-Fensham (1992)	20*-Dickinson & Kirkpatrick (1987)	12-van Etten (1995)
0.25*-Dickinson & Kirkpatrick (1987); Brown *et al.* (1991); Williams *et al.* (1995)	25-Finlayson *et al.* (1993)	15-Fensham (1992)
0.5-Raison *et al.* (1983); Bowman & Wilson (1988); Crockford & Richardson (1995)		20-Bowman & Wilson (1988)
1.0-Finlayson *et al.* (1993)		

Fuel modelling, like any modelling procedure, has its limitations. For example, the highly discontinuous nature of the fuel bed in most vegetation types can undermine the predictive ability of some models (McCaw 1988). McCaw (1988) went further and added that empirically-derived models are only useful under the conditions for which they were derived, and that models from first principles were impractical unless accompanied by simple techniques for estimating their parameters. Despite these and other potential limitations, much research effort has been directed at fuel dynamics modelling. This is because, of the three variables that control fire intensity (climate, topography and fuel load), the quantity of available fuel is the most amenable to modification by management practices (Sandercoe 1986).

Fuel loads are therefore of considerable interest to management authorities. A knowledge of fuel loads within a given area is essential for the planning and ignition of prescribed burning programs and the formulation of overall fire management plans (Marsden-Smedley & Catchpole 1995). Further, an understanding of fuel loads is a critical component of the minimum data set required for the effective monitoring of fire on conservation reserves (Gill & Nichols 1989; Wardell-Johnson *et al.* 1989; Skull 1993; Warriner-Lawrence 1993).

While the advantages and disadvantages of prescribed burning for conservation and hazard reduction are still widely debated, all of Australia's conservation management authorities currently have fuel reduction burning programs in place. These programs range from the very sophisticated, involving high-powered computer modelling, to the very basic where fuel loads are estimated visually. Large GIS-based programs have been developed for specific national parks based on models from the United States (e.g. PREPLAN in Kosciusko National Park, Kessel *et al.* 1984; Kessel 1990) where fuel accumulation curves are constructed based on some field harvest data and vegetation structure. Others have been developed to investigate the economics of prescribed burning for rural land managers (e.g. BURNECON, MacLeod & Ludwig 1991). More recently, Wildfire Threat Analysis has been developed to examine, assess and map potential wildfire hazards in conservation reserves (Malcolm *et al.* 1995; Vodopier & Haswell 1995). This assessment is based on hazards (of which fuel loads are a component), risks (the potential for ignition) and values (the value of the asset which may be destroyed) (Malcolm *et al.* 1995).

A brief review of fuel loads documented for plant communities within the wet-dry tropics of Australia is provided by Gill *et al.* (1991), with most grassy woodlands in the region having fuel loads of 5-10 t ha^{-1} four to seven years after fire. Finlayson *et al.* (1993) provide the only published example of fuel loads in *Melaleuca* forests for northern Australia. In a study on the Magela floodplain near Jabiru in the Northern Territory, they reported fuel loads ranging from 5-21 t ha^{-1} beneath a mixed forest of *M. viridiflora* and *Melaleuca cajuputi* Powell. Enormous variation in the fuel loads was documented which was partially related to the small sample size used, but also attributed to the inherently large spatial variability of fuel at the site. Additionally, the pattern of water flow and litter transport across the site in the wet season were considered important factors accounting for the high level of variability (Finlayson *et al.* 1993). The authors concluded that regardless of the variability, the average fuel loads recorded (11 t ha^{-1}) were higher than those documented for tropical rainforests (Spain 1984), indicating the highly productive nature of these forests and their important role in the detrital/debris turnover cycle on the floodplain.

5.1.3 Fire temperature determination

For the past 50 years, fire researchers have been interested in determining the temperatures reached in fires. Primarily, this is because the temperature reached during a fire can be the definitive cause of plant tissue death (Whelan 1995). Beadle (1940) used organic compounds with different melting points contained within glass tubes to investigate soil temperatures in forest fires in NSW. Temperature-sensitive paints and crayons have also been widely used (Smith & Sparling 1966; Sparling & Smith 1966; Hobbs *et al.* 1984), although problems with their calibration have been identified (Gill & Knight 1991). These problems stemmed from the fact that they were manufactured to change colour only after exposure to the pre-determined temperature for long periods (up to 30 minutes). As Gill and Knight (1991) correctly commented, residence times in fires are usually measured in terms of seconds rather than minutes.

Omega International Inc. have now developed a range of temperature-sensitive products that are accurate to within 0.1% of the quoted temperature, and have response times measured in milliseconds. These products are inexpensive, produce reliable results over a broad range of temperatures, are simple to use and calibrate well with the manufacturer's indications (Gibson *et al.* 1990). With the advent of portable dataloggers, and advances in sensor research and development, thermocouples are now also widely used (Raison *et al.* 1986; Bradstock *et al.* 1992;). These electronic devices are not without their own limitations, and for a more comprehensive review of these methods, the reader is referred to Gill and Knight (1991).

5.1.4 Aims of this investigation

This section of the thesis had the following objectives:

(i) to investigate the temperatures reached during a prescribed fire in *M. viridiflora* woodlands;

(ii) to assess fine fuel load accumulation in *M. viridiflora* woodlands and a tropical lowland eucalypt community, following a fire event;

(iii) to investigate the effects of single fires (prescribed and wildfires) on the structural characteristics of *M. viridiflora* woodlands; and

(iv) to simulate the effects of repeated burns on regeneration of the midstorey in *M. viridiflora* woodlands using a prescribed burn and an artificially induced fire treatment. This experiment was designed to test the hypothesis that annual burning of these communities was not ecologically sustainable.

5.2 METHODS

5.2.1 Fire temperature determination

To determine the likely temperatures reached during fire in a *M. viridiflora* community, a temperature trial experiment was conducted at Site 5 on HINP. In this experiment, *Omegalaq* temperature-indicating liquids (60°C, 100°C, 200°C, 300°C, 450°C and 600°C) were painted onto the centre of one half of a 20 x 10 cm sheet of aluminium. These liquids are coloured and become clear when their calibrated temperature is reached. The sheets were then folded in half and closed over completely with a rubber mallet. This prevents flames from coming into direct contact with the paints and discolouring them. In the field, 20 sheets were placed between the litter layer and the soil. Ten sheets were placed in areas with relatively high fuel loads (estimated visually), and another ten in regions with lower fuel loads. Following a small-scale experimental fire, the plates were re-opened, and the temperature range that the fire reached at each location determined. Prior to this experiment, these paints had not been tested in Australian plant communities (Mr K. Tolhurst, pers. comm.).

5.2.2 Structural changes following fire

Site 5 (HINP) was selected for detailed investigation as it was the focus of QDEH fire management objectives for this vegetation type (QDEH 1995b), it formed a strategic base from which to access the other HINP sites, it was part of the largest tract of this vegetation type within HINP (occurring as large, convoluted, interconnected patches as well as small isolated patches) and its recent fire history was well documented. During the course of this study, it became possible to include three additional sites as other fire management studies proceeded. These included:

- Site 6 (Chapter 3, Table 3.4) - A late dry season burn (October 1993) was conducted in preparation for a larger, more intense prescribed management burn on HINP that followed at Site E (described in Section 5.2.4). Sampling was conducted between October 1993 and June 1995. Further details of all the fires referred to in this thesis are presented in Appendix E.
- Site 14 (Table 3.4) - This site was burnt in October 1995 as part of the investigation into exotic species invasion of *M. viridiflora* woodlands outlined in Chapter 6 (Section 6.1). Data were collected twice only at this site (immediately prior to and after fire) from three parallel transects (each 400 m x 10 m, therefore the total area equalled 12,000 m^2) used in the exotic species study, rather than the smaller area of the actual site plots (500 m^2).
- Site 19 (Table 3.4) - This site was accidentally burnt in October 1993 by Queensland Forest Service staff as part of a backburn around the adjacent pine plantation. Data were collected twice before the fire (July 1992 & August 1993) and then periodically following the fire until June 1995.

At each site, changes to the following community attributes were recorded:

- abundances of *M. viridiflora* in two main size classes. The first of these were the spouts and small saplings at ground level (<1.5 m) and the second involved the midstorey which was defined as all individuals with heights in the range 1.5 to 3.0 m; and

- density of the dominant understorey species, *X. johnsonii.*

Additionally, at two sites (Site 5 and 10) the recruitment of seedlings following fire was monitored. Twenty randomly located, permanently marked 50 cm^2 quadrats were utilised to record seedling density from July 1992 until June 1995.

5.2.3 Simulated fire experiment

To determine the effects of repeated fires on the regeneration of the midstorey (potentially the next canopy tree cohort) of *M. viridiflora* woodlands, an experiment was established at Site 5 following the prescribed fire of September 1992 (see Appendix E for details). Forty *M. viridiflora* individuals within the 1-3 m size class were randomly selected and then randomly assigned to one of the following treatments:

- plants subject to a single fire (the prescribed fire) in a two year period; and
- plants subject to an annual fire regime, i.e. the prescribed fire and a simulated fire 12 months later (October 1993).

The selected plants were labelled with aluminium tags. The height of the major stem and all coppice shoots were recorded over an eighteen month period that began 12 months after the prescribed fire. Data were collected on six occasions for the number of coppice shoots and five occasions for the length of coppice shoots. This change in sampling frequency occurred because, although it was possible to identify the number of emerging coppice shoots on the first sampling occasion, it was not possible to measure them. A t-test prior to the initiation of the experiment confirmed that there were no significant differences in either the number or length of coppice shoots on each plant in the different treatments ($p=0.32$ and $p=0.24$ respectively).

To simulate a prescribed burn, a small *Primus* blow torch was used, connected to a portable 0.34 kg gas cylinder. Prior to use in the field, the temperature of the flame was measured at varying distances from the tip of the blow torch using a calibrated thermocouple with a range of 0-1000°C. Knowing that the temperatures recorded in a prescribed fire ranged from less than 60°C to more than 600°C, with the majority of temperatures in between the less than 60°C category and 300°C (Section 5.3.1), a temperature of 200°C was selected for use in the simulated fire experiment. This temperature was recorded at a distance of 20 cm from the tip of the blow torch and the treatment duration of 10 seconds was selected following consultation with a leading Australian fire researcher (Dr M. Gill, pers. comm.). The response of plants in both treatments was then documented (heights and number of coppice shoots) on six occasions over a 20 month period, and analysed using a repeated measures ANOVA. SAS Version 6.0 was used for the repeated measure ANOVAs and SPPS Version 6.0 for all other data analysis procedures.

5.2.4 Fuel load accumulation

Fuel loads were investigated between July 1991 and June 1995 at Site 5 (see Chapter 3, Table 3.4 for detailed site information). Two other sites were selected for investigation, namely:

- Site 6 (rationale outlined in Section 5.2.2) with sampling also conducted between July 1992 and June 1995; and
- Site E - A very tall open forest site (classified using Walker & Hopkins 1990) dominated by *Eucalyptus cloeziana* F. Muell. (Gympie messmate). Little of this habitat remains on alluvial soils in the WTBR and the site has recently (< 50 years) been invaded by rainforest species (Mr P. Stanton, pers. comm.). The QDEH management aim at this site was to utilise a hot fire (Appendix E) to regenerate the *E. cloeziana* open forest without the rainforest element (QDEH 1995c). This site was investigated to allow comparisons to be made with other studies conducted on eucalypt-dominated communities within Australia. Sampling took place between August 1991 and June 1995.

At all sites a 0.25 m^2 (0.5 x 0.5 m) quadrat was used to sample all live and dead fuel both on the ground and up to a height of 1.5 m. This three dimensional "fuel block" represents the fuel that should burn in a moderately intense fire (Mr P. Stanton, pers.

comm.). Large woody stems and branches (> 20 mm diameter) that are unlikely to ignite in low to moderate intensity fires were excluded from the sampling procedure. This method is similar to that used by Sandercoe (1986) in an investigation of the fuel dynamics of Cooloola National Park.

Ten randomly-located samples were collected at each site. Samples were then transferred to the laboratory soon after harvest and oven-dried for 24 hours at 80°C or until no further weight loss was recorded. The samples were then weighed to determine dry weights for both the live and dead components. Results were then averaged for each site, with all weights expressed in tonnes per hectare (t ha^{-1}). Scorch heights were measured in the field to provide an estimate of fire intensity. In the absence of more sophisticated techniques, this method has been used successfully to aid in the classification of fire intensity (e.g. Christensen *et al.* 1981).

To compare the results of this study with others that have documented exponential rises to a maximum, an exponential curve was fitted to the data. This is possible within the Sigmaplot for Windows (Version 2.0) graphics package using the exprise.fit file in the curve-fit equation (sample files) section of the program.

5.3 RESULTS

5.3.1 Fire temperature determination

The temperatures recorded by the heat sensitive paints are summarised in Table 5.4. A wide range of temperatures were recorded indicating a patchy burn. Although the modal temperature range was less than 60 °C, three plates recorded temperatures in excess of 600 °C . The data were primarily obtained to allow selection of a suitable temperature for the simulated fire experiment (see Section 5.3.3) and therefore will not be discussed further.

Table 5.4 **Temperatures (°C) recorded during a small-scale experimental fire in *M. viridiflora* woodland (Site 5).** Note that no temperatures in either the 60-100°C or 450-600°C range were recorded.

Temp. (°C)	<60	100-200	200-300	300-450	>600
No. of plates	9	4	1	3	3

5.3.2 Structural changes following fire

Plates 2a-f document the changes following fire at Site 5 on HINP. Immediately following the fire (Plate 2c), all available fine fuel had been consumed (see Section 5.3.4 for details). Scorch heights indicated a fire of moderate intensity. Care must be taken not to over-estimate fire intensity with *M. viridiflora* as the bark burns readily and therefore records higher scorch marks than other species including many of the eucalypts. Two months after fire, regeneration was initiated from epicormic buds and coppice shoots (*M. viridiflora*), and protected apical buds (*X. johnsonii*) (Plate 2d). Six months after fire more widespread regeneration was observed and fuel began to re-accumulate in the understorey (Plate 2e). Thirty months after fire, although the fire scars were still evident, regeneration was well advanced (Plate 2f).

Prior to the fire, the first two recordings (July 1993 and August 1993) of the density of *M. viridiflora* sprouts in the understorey were similar (Sites 6 and 19) (Figures 5.1a-c). Following fire, however, density increased dramatically. This increase was not sustained over time, except at Site 5 where a second major rise occurred at the end of the sampling program (Figure 5.1a). Site 19 exhibited the largest increase, with density in the understorey rising from 276 per 500 m^{-2} in July 1992 to more than 900 in December 1993 just two months after fire (Figure 5.1c). The effect of fire on the individuals in the midstorey of each site is presented in Figure 5.2. Losses in the 1.5 to 3.0 m height category ranged from 38% at Site 6 to nearly 60% at Site 14. The mean loss following fire over all sites was almost half (49%).

Plate 2 **Time series of photographs showing the effects of, and recovery from, a prescribed fire (30.9.92) at study Site 5 on Hinchinbrook Island National Park.**

(a) Site 5: Pre-fire (July 1992).

(b) Site 5: Aerial view of prescribed burn.

(c) Site 5: Two days after fire.

(d) Site 5: Two months after fire.

(e) Site 5: Six months after fire.

(f) Site 5: 30 months after fire.

Figure 5.1 **Density of *M. viridiflora* shoots (500 m^{-2}) in the understorey of Sites 5, 6 (Hinchinbrook Island National Park) and 19 (mainland) prior to and following fire.** Full details of the fires are presented in Appendix E.

(a) Site 5 - burnt in September 1992.

(b) Site 6 - burnt in October 1993.

(c) Site 19 - burnt prior to October 1993.

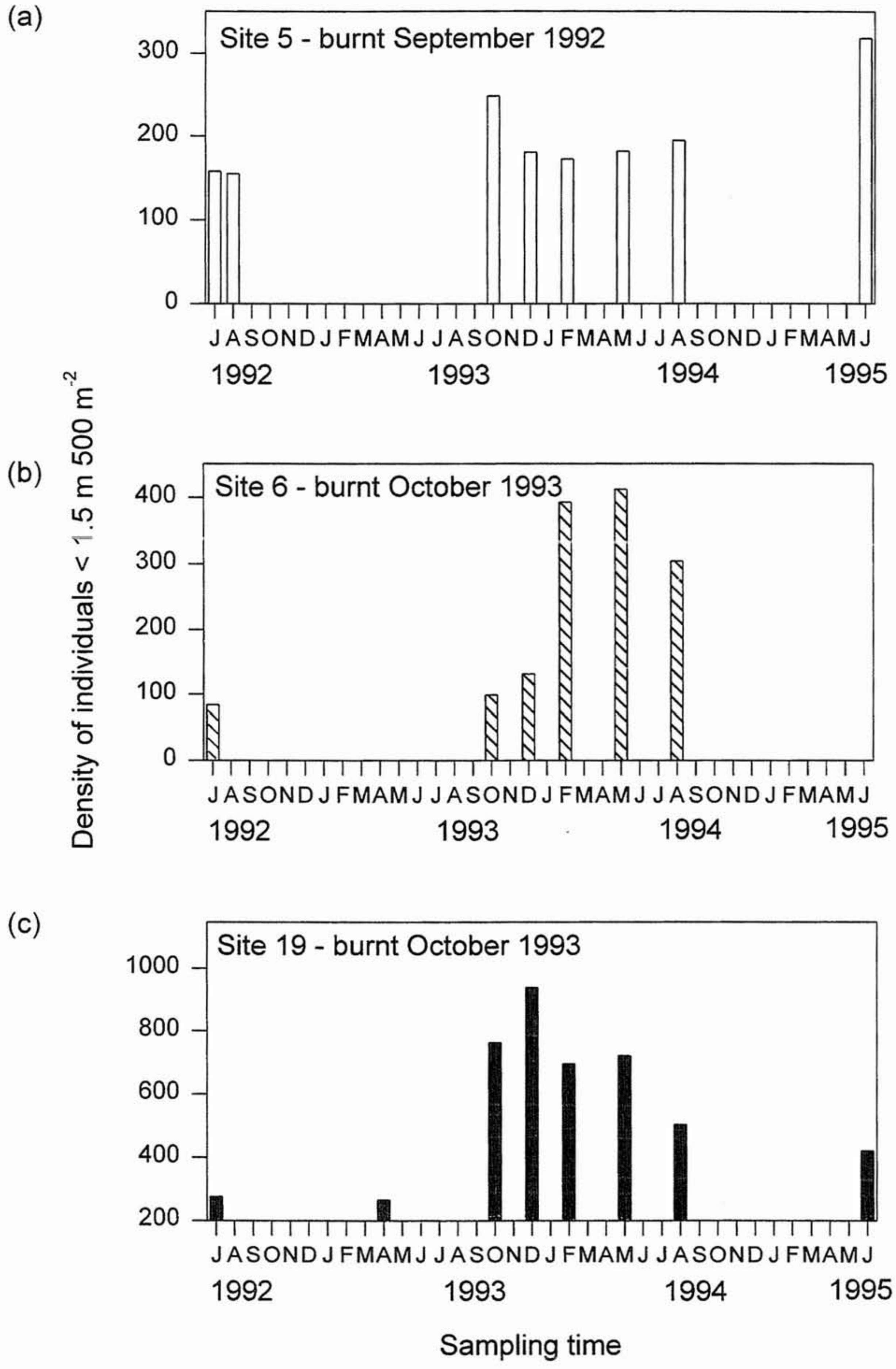
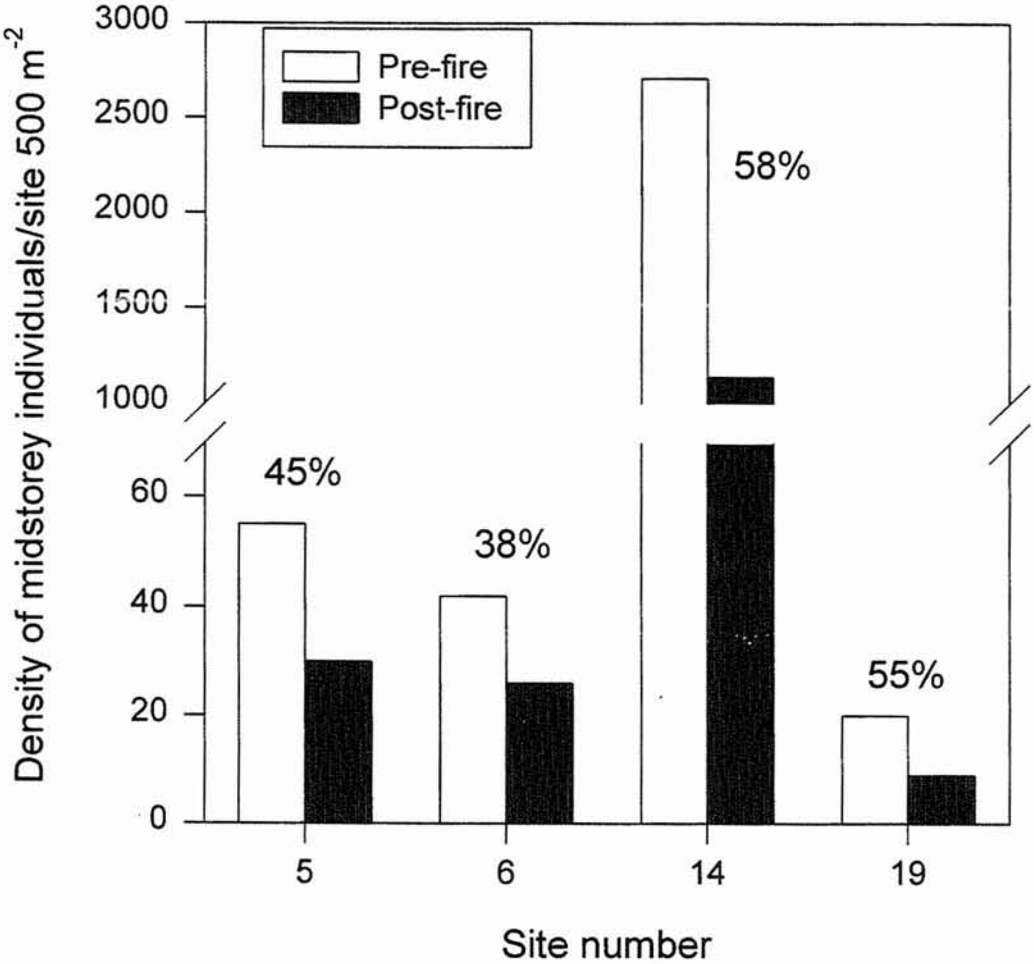

(a)
Site 5 - burnt September 1992
300
200
100
0
J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1992
1993
1994
1995
(b)
Site 6 - burnt October 1993
400
300
200
100
0
J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1992
1993
1994
1995
(c)
Site 19 - burnt October 1993
1000
800
600
400
200
J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1992
1993
1994
1995
Density of individuals < 1.5 m 500 m-2
Sampling time

Figure 5.2 **Density and the percentage reduction (%) of individuals in the 1.5 to 3.0 m height category at Sites 5, 6 (Hinchinbrook Island National Park) and Site 19 (mainland) prior to and following fire.** Sites 5, 6, & 19 - 500 m^{-2} and Site 14 - 12,000 m^{-2}). Full details of the fires are presented in Appendix E.


Density of midstorey individuals/site 500 m-2
3000
2500
2000
1500
1000
60
40
20
0
Pre-fire
Post-fire
58%
45%
38%
55%
5
6
14
19
Site number

Before the fires, the density of *X. johnsonii* in the understorey of each community was similar for the sampling times assessed (July 1992 and October 1993). As with the shoots of *M. viridiflora*, however, a steady rise was recorded following fire (Figures 5.3a-c). The largest increase occurred at Site 6 where 20 months after fire the density had increased by 231% (Figure 5.3b). Site 5 was the only site to exhibit any decline in *X. johnsonii* density, which only occurred at the final recording nearly three years after fire (Figure 5.3a).

The recruitment of *M. viridiflora* seedlings at Sites 5 and 10 is presented in Figure 5.4. Marked increases in recruitment were evident following fire at both sites. At Site 10, seedling density tapered off gradually until near pre-fire levels were recorded in the final (June 1995) sample. Seedlings at Site 5 were very short-lived, with a marked increase in mortality recorded less than 18 months after fire.

5.3.3 Simulated fire experiment

Between October 1993 and August 1994 (five sampling events), plants subject to two fires had higher numbers of coppice shoots than those subjected to a single fire (Figure 5.5a). This was particularly evident in the February 1994 and August 1994 samples with the trend reversed in the final sample taken 20 months after the second (simulated) fire. The repeated measures ANOVA indicated a significant time and time*treatment effect, although the time effect was less pronounced (Table 5.5). The uni-variate tests calculated as part of the ANOVA process confirmed these findings.

The mean length of coppice shoots was three to four times higher for individuals in the single fire treatment, except for the final sample when they were approximately twice the length (Figure 5.5b). In June 1995 plants in the single fire treatment showed a marked decline in mean coppice shoot length whilst the repeated fire treatment exhibited a slight increase from the previous sample. As for the repeated measures ANOVA on the number of coppice shoots, a significant time and time*treatment effect were recorded for the coppice shoot length data analysis, again with the time effect less pronounced (Table 5.6). In contrast, however, the uni-variate time effect was not significant ($p=0.14$). An examination of sphericity for both repeated measures indicated that the assumptions of both ANOVAs were not violated during the analysis.

Figure 5.3 **Density of *X. johnsonii* (500 m^{-2}) in the understorey of Sites 5, 6 (Hinchinbrook Island National Park) and 19 (mainland) prior to and following fire.** Full details of the fires are presented in Appendix E.

(a) Site 5 - burnt in September 1992.

(b) Site 6 - burnt in October 1993.

(c) Site 19 - burnt prior to October 1993.

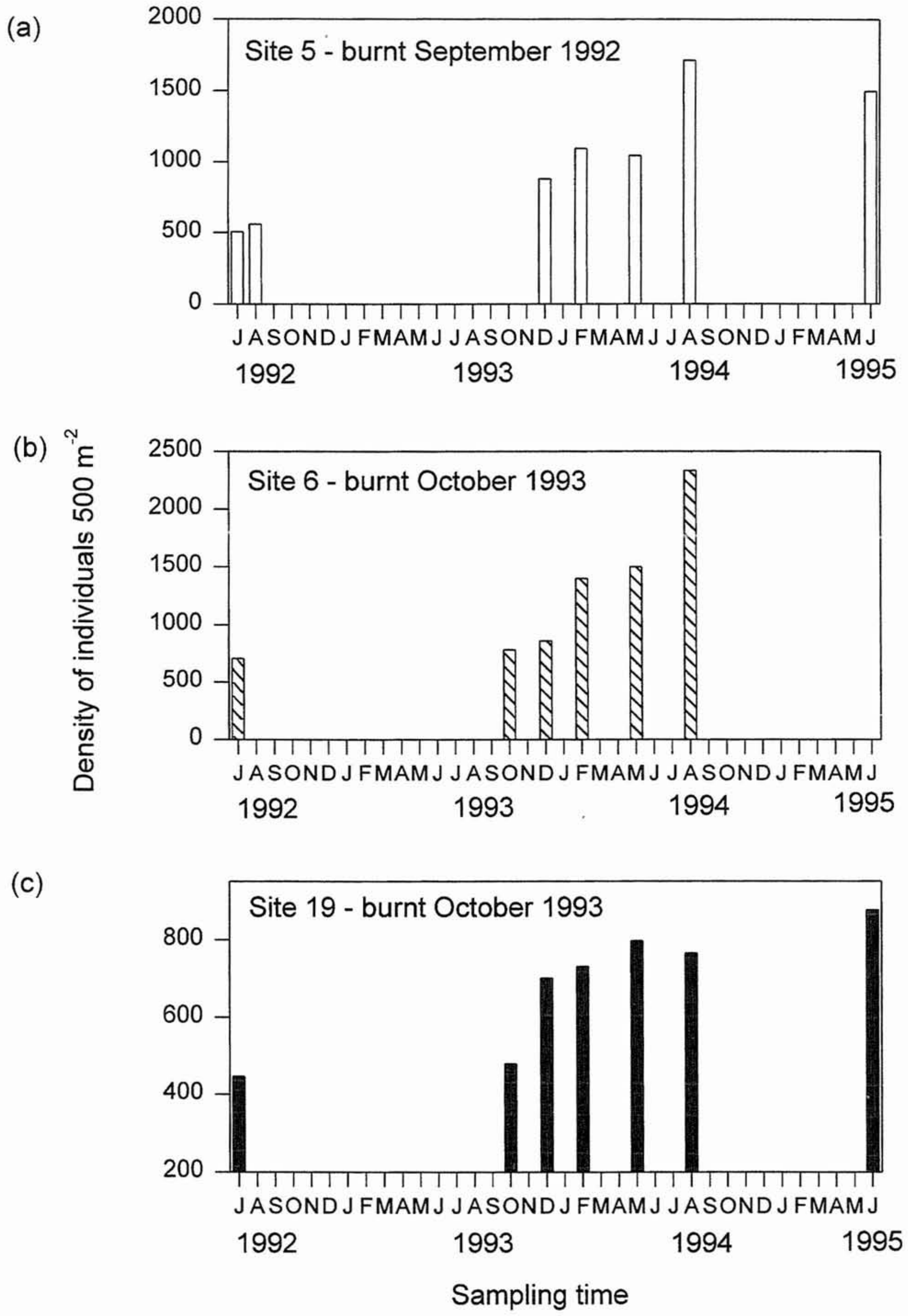

(a)
Site 5 - burnt September 1992
2000
1500
1000
500
0
J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1992
1993
1994
1995
(b)
Density of individuals 500 m^{-2}
Site 6 - burnt October 1993
2500
2000
1500
1000
500
0
J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1992
1993
1994
1995
(c)
Site 19 - burnt October 1993
800
600
400
200
J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1992
1993
1994
1995
Sampling time

Figure 5.4 **Pattern of *M. viridiflora* seedling survival (mean density of seedlings 0.25 m^{-2}) in the understorey of Sites 5 and 10 (Hinchinbrook Island National Park).** Bars indicate standard errors.

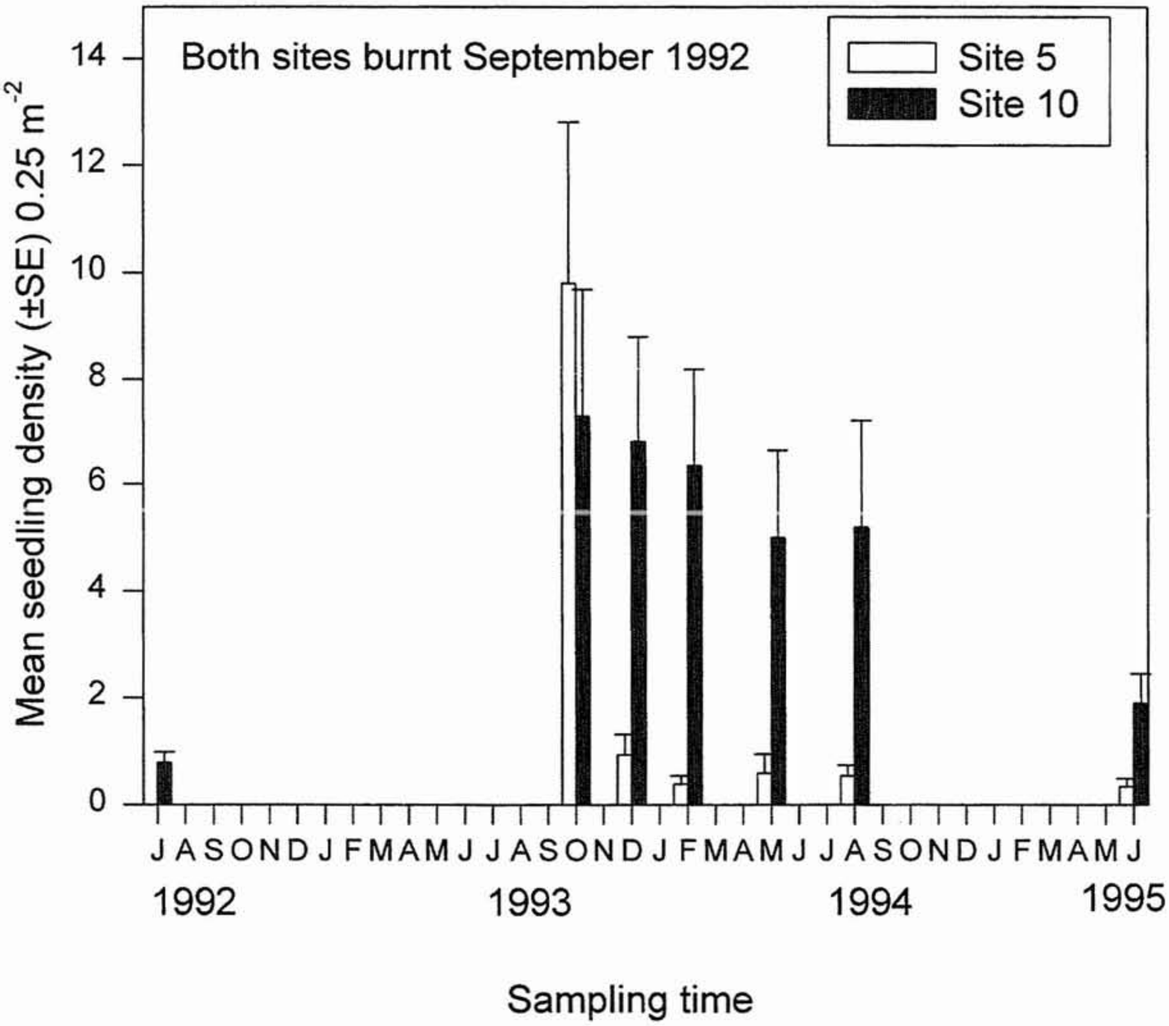

Both sites burnt September 1992
Site 5
Site 10
Mean seedling density (±SE) 0.25 m^{-2}
0
2
4
6
8
10
12
14
J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1992
1993
1994
1995
Sampling time

Figure 5.5 **Growth responses of *M. viridiflora* midstorey (1.5 to 3.0 m) individuals to single (September 1992) and repeated (2) fires (September 1992 and October 1993) at Site 5 on Hinchinbrook Island National Park.** Bars indicate standard errors. Full details of the fires are presented in Appendix E.

(a) **Mean number of coppice shoots per plant following a single burn and repeated (2) burns.**

(b) **Mean coppice shoot length following a single burn and repeated (2) burns.**

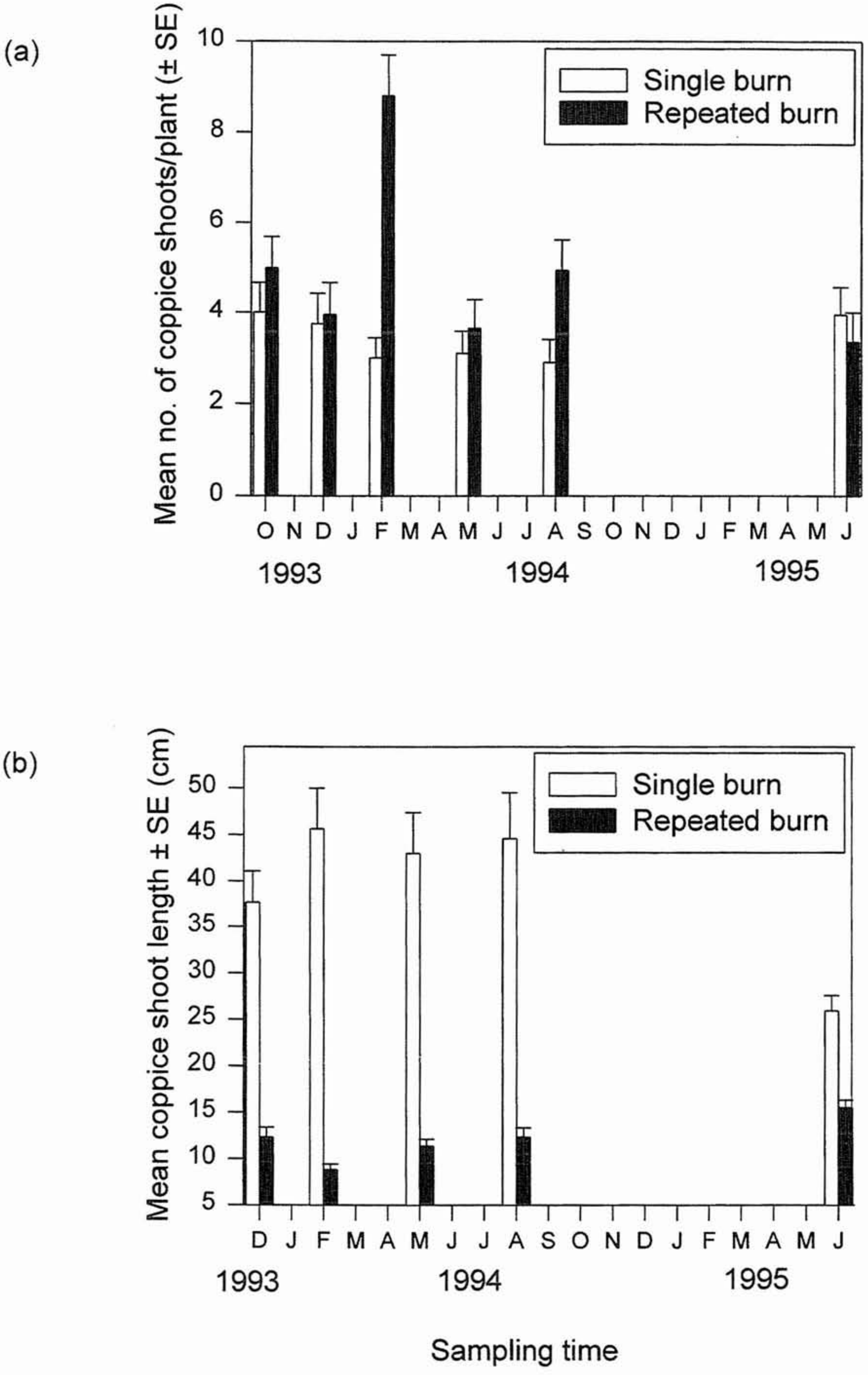

(a)
Mean no. of coppice shoots/plant (± SE)
0
2
4
6
8
10
Single burn
Repeated burn
O N D J F M A M J J A S O N D J F M A M J
1993
1994
1995
(b)
Mean coppice shoot length ± SE (cm)
5
10
15
20
25
30
35
40
45
50
Single burn
Repeated burn
D J F M A M J J A S O N D J F M A M J
1993
1994
1995
Sampling time

Table 5.5 **Results from the repeated measures ANOVA of the number of coppice shoots in the two treatments (single burn vs. repeated burns) over the duration of the experiment.** Significant results are highlighted in bold.

Variable/Interaction	F ratio	df	p value
Multi-variate			
Time	7.01	5	**<0.0001**
Time*treatment	13.53	5	**<0.0001**
Between subjects	4.59	1	**0.04**
Uni-variate			
Within subjects (time)	7.84	5	**<0.0001**
Within subjects (time*treatment)	12.27	5	**<0.0001**

Table 5.6 **Results from the repeated measures ANOVA of the length of coppice shoots in the two treatments (single burn vs. repeated burns) over the duration of the experiment.** Significant results are highlighted in bold.

Variable/Interaction	F ratio	df	p value
Multi-variate			
Time	3.39	4	**<0.01**
Time*treatment	9.70	4	**<0.0001**
Between subjects	225.94	1	**<0.0001**
Uni-variate			
Within subjects (time)	1.74	4	0.14
Within subjects (time*treatment)	5.93	4	**<0.0001**

5.3.4 Fuel load accumulation

The percentage fuel reduction and the fuel load 18 months after fire (also expressed as a percentage of the pre-fire fuel load) are summarised for all sites in Table 5.7. The more intense fires (indicated by higher scorch heights) resulted in a greater immediate reduction in the amount of available fuel.

At Sites 5 and 6 dead fuel was more abundant in the pre-fire samples (Figures 5.6 and 5.7 respectively). This was also the case at Site 14 where pre-fire fuel load totals for the dead and live components were 6.6 and 0.16 t ha^{-1} respectively. Following fire at the *M. viridiflora* sites the live component of the fuel became dominant. Dead fuel was always the most abundant component at the eucalypt forest site (Site E, Figure 5.8). At all sites fuel loads remained substantially below pre-fire levels up until at least 18 months after fire, and at Site 5 for almost three years after fire (Figure 5.6). More specific comments for each site are considered below. Although no data on litterfall (which is a component of most fuel accumulation models) were collected during this study, an exponential curve (recall Olson's (1963) fuel accumulation model was based on an exponential curve) was fitted to the empirical data for Sites 5, 6, and E (Figures 5.9a-c).

Table 5.7 **Fine fuel load accumulation and general fire data.** The number in parentheses for scorch heights indicates the maximum value recorded. Previous fire dates provided from QDEH and QSFS records.

Site	% fine fuel reduction after one month	Accumulation as a % of original fuel load after 18 months (%)	Scorch height range (m)	Previous fire
5	95	64	2-3 (4)	1983
6	89	35	1-2 (3)	197?
14	97	n/a	4-5 (6)	1986
E	84	49	0.5-1 (3.5)	197?

Figure 5.6 **Pattern of fuel load accumulation (t ha^{-1}) following fire (September 1992) in a *M. viridiflora* woodland at Site 5 on Hinchinbrook Island National Park.** Where appropriate, bars indicate standard errors. Full details of the fire are presented in Appendix E.

(a) Live material.

(b) Dead material.

(c) Total.

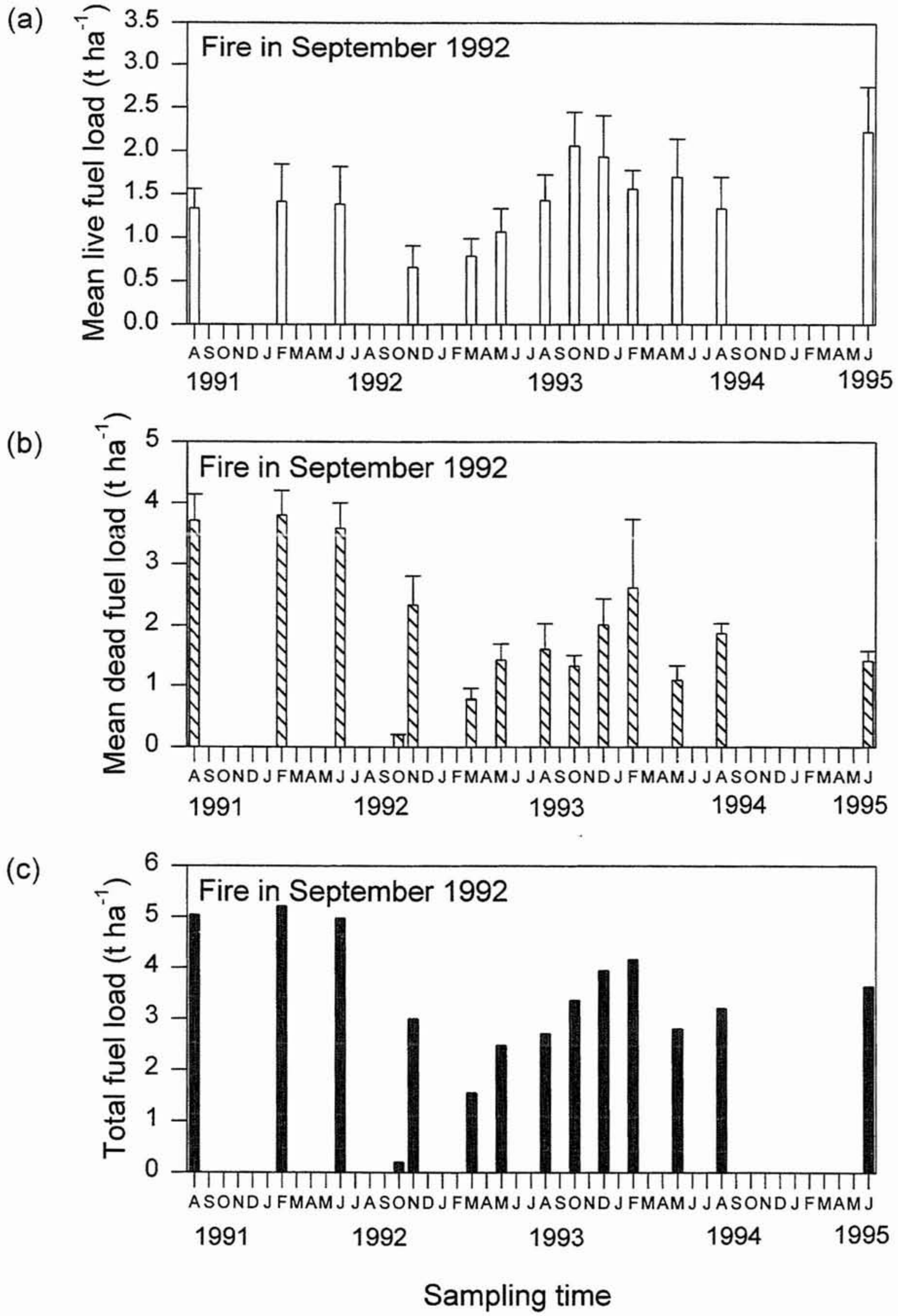

(a)
Mean live fuel load (t ha^{-1})
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
Fire in September 1992
A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1991
1992
1993
1994
1995
(b)
Mean dead fuel load (t ha^{-1})
0
1
2
3
4
5
Fire in September 1992
A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1991
1992
1993
1994
1995
(c)
Total fuel load (t ha^{-1})
0
1
2
3
4
5
6
Fire in September 1992
A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1991
1992
1993
1994
1995
Sampling time

Figure 5.7 **Pattern of fuel load accumulation (t ha^{-1}) following fire (October 1993) in a *M. viridiflora* woodland at Site 6 on Hinchinbrook Island National Park.** Where appropriate, bars indicate standard errors. Full details of the fire are presented in Appendix E.

(a) Live material.

(b) Dead material.

(c) Total.

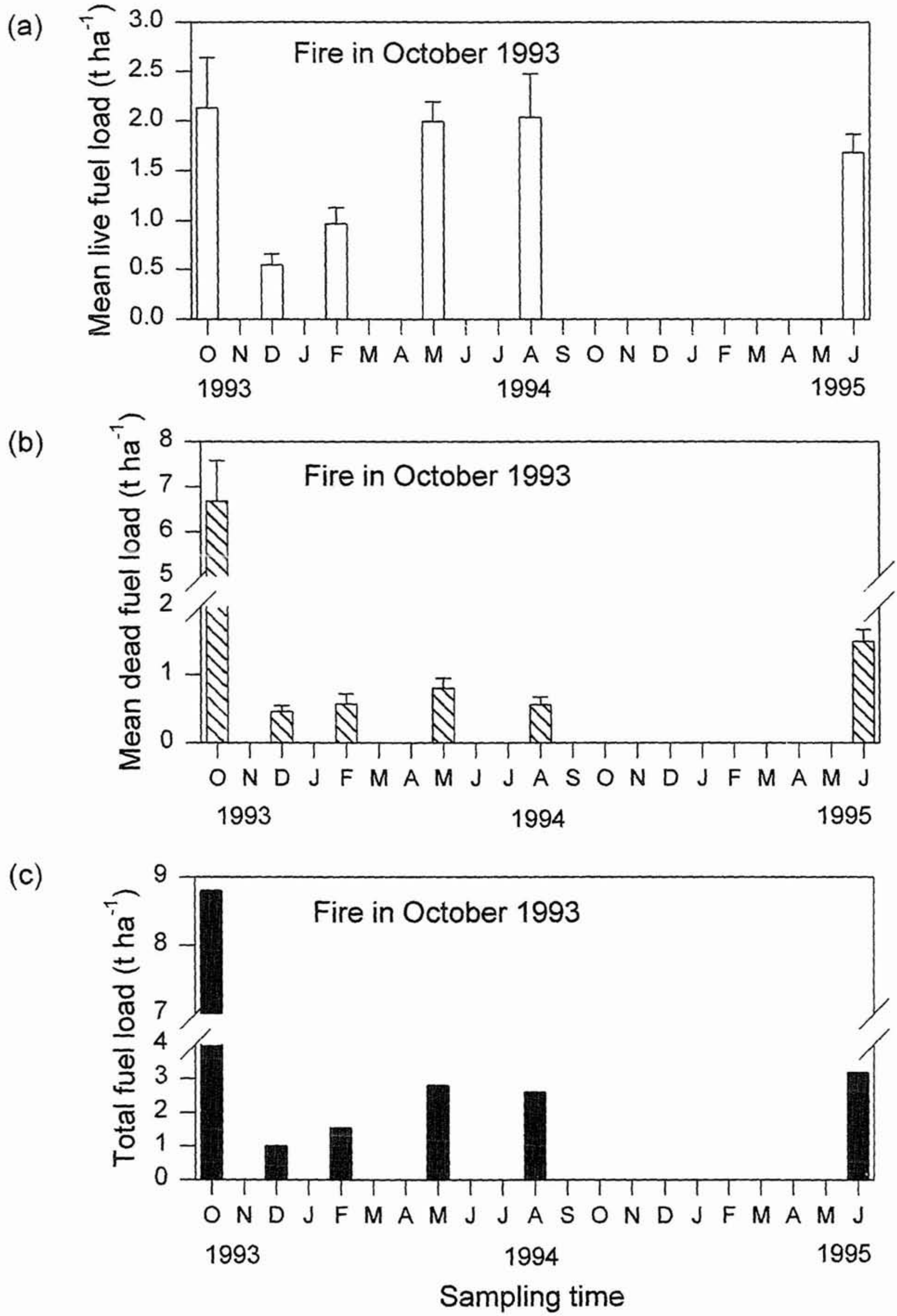
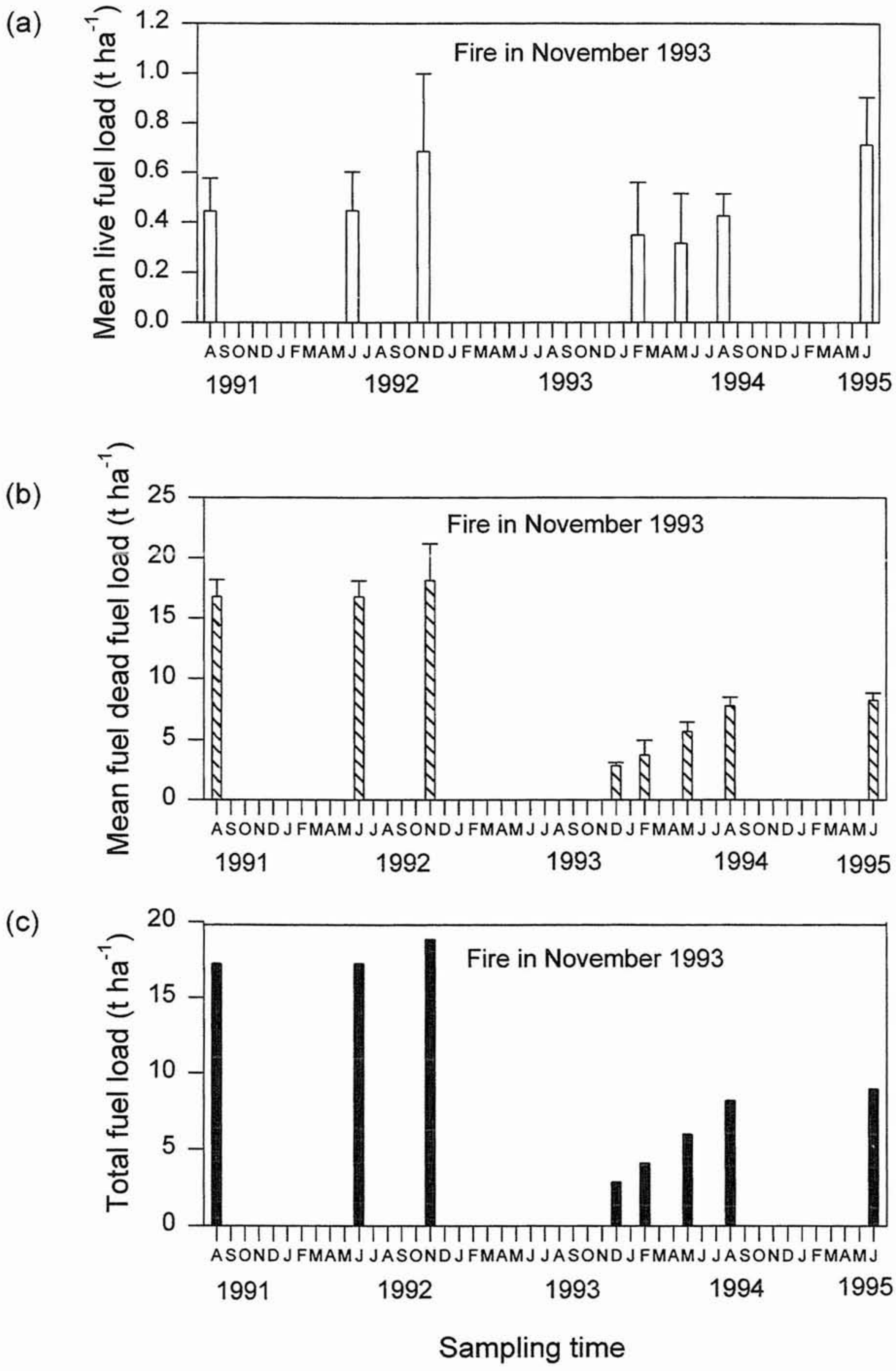

(a)
Mean live fuel load (t ha^{-1})
3.0
2.5
2.0
1.5
1.0
0.5
0.0
Fire in October 1993
O N D J F M A M J J A S O N D J F M A M J
1993
1994
1995
(b)
Mean dead fuel load (t ha^{-1})
8
7
6
5
2
1
0
Fire in October 1993
O N D J F M A M J J A S O N D J F M A M J
1993
1994
1995
(c)
Total fuel load (t ha^{-1})
9
8
7
4
3
2
1
0
Fire in October 1993
O N D J F M A M J J A S O N D J F M A M J
1993
1994
1995
Sampling time

Figure 5.8 **Pattern of fuel load accumulation (t ha^{-1}) following fire (November 1993) in a *E. cloeziana* forest at Site E on Hinchinbrook Island National Park.** Where appropriate, bars indicate standard errors. Full details of the fire are presented in Appendix E.

(a) Live material.
(b) Dead material.
(c) Total.


(a)
Mean live fuel load (t ha^{-1})
1.2
1.0
0.8
0.6
0.4
0.2
0.0
Fire in November 1993
A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1991
1992
1993
1994
1995
(b)
Mean fuel dead fuel load (t ha^{-1})
25
20
15
10
5
0
Fire in November 1993
A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1991
1992
1993
1994
1995
(c)
Total fuel load (t ha^{-1})
20
15
10
5
0
Fire in November 1993
A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J
1991
1992
1993
1994
1995
Sampling time

Following the large reduction in available fuel at Site 5 on HINP in October 1992, a rapid rise in dead material was recorded, possibly due to a time-lag effect (late senescence of tree litter) associated with the fire (Figure 5.6). A rapid rise in total fuel load was then recorded until the May 1994 sample, when a large decrease of over 1 t ha^{-1} occurred (mostly attributable to the dead fuel component). As a result of these large fluctuations the fit of the exponential curve to the empirical data is not close (Figure 5.9a).

Despite being subjected to a fire of lower intensity, a large reduction in available fuel also occurred at Site 6 (Figure 5.7). The live component of the fuel load sample returned to pre-fire levels within six months (October 1993 sample), whilst the dead component exhibited a very slow pattern of re-accumulation. The exponential curve shows a good fit with the empirical data (Figure 5.9b).

At Site E (dominated by *E. cloeziana*) the live component of the fuel load sample increased rapidly following an initial lag after burning in November 1993 (0 t ha^{-1} for December 1993 sample, Figure 5.8). The dead fuel increased steadily following fire. This re-accumulation pattern, however, appeared to have stalled (at least temporarily) as indicated by the June 1995 sample. The exponential curve fits the empirical data closely (Figure 5.9c).

5.4 DISCUSSION

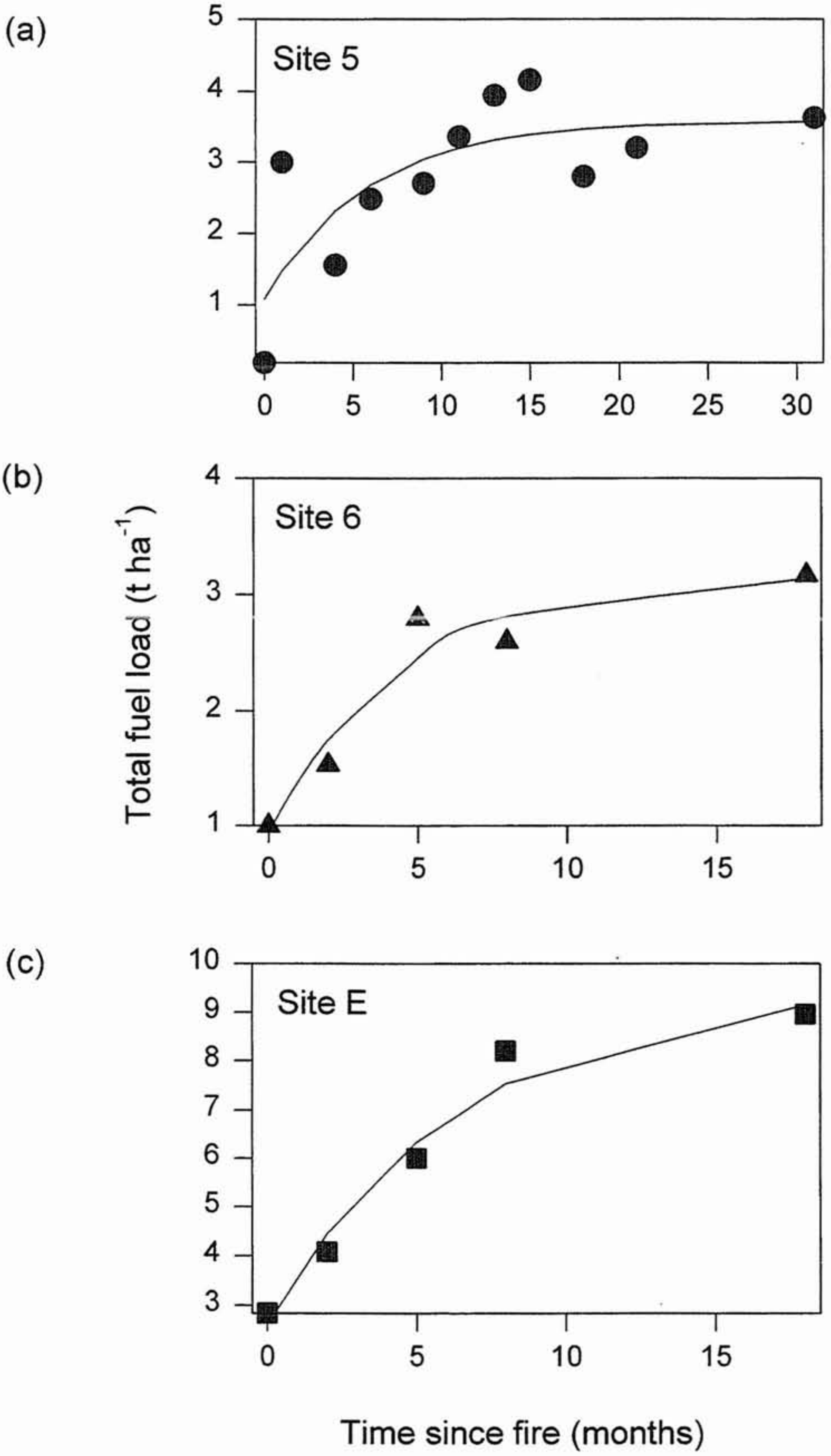
Melaleuca viridiflora woodlands are considered well adapted to fire (Stocker & Mott 1981), as is the species itself because of its ability to coppice profusely (Tweddell 1982). This study has shown that even moderate intensity fires have pronounced effects on these communities, resulting in an average loss of 49% of individual trees in the 1.5 to 3.0 m height class (Figure 5.1). When subjected to a second fire 12 months later, these individuals produced higher numbers of coppice shoots with slower growth rates than individuals subject to a reduced fire frequency (Figure 5.5a-b). The end result of repetitive fires appears to be an individual reduced to little more than a profusely coppicing lignotuber, the resprouting ability of which may be exhausted when subjected

Figure 5.9 **Exponential curve {$f(x) = ae^{-bx} + ce^{-dx}$} fitted to empirical total fuel load accumulation (t ha^{-1}) data at three sites (Sites 5, 6 and E) on Hinchinbrook Island National Park.** Full details of the fires are presented in Appendix E.

(a) **Site 5 - *M. viridiflora* woodland.**

(b) **Site 6 - *M. viridiflora* woodland.**

(c) **Site E - *E. cloeziana* forest.**


(a)
Site 5
5
4
3
2
1
0
5
10
15
20
25
30
(b)
Site 6
4
3
2
1
0
5
10
15
Total fuel load (t ha^{-1})
(c)
Site E
10
9
8
7
6
5
4
3
0
5
10
15
Time since fire (months)

to a recurring annual fire regime in this region. It should be noted that the success of lignotuberous sprouting following a fire of given intensity and duration depends upon the depth of soil above the organ and its stage of development (Whelan 1995). An assessment of the importance of these variables to the regeneration of *M. viridiflora* could form the basis for future research.

In terms of overall community structure, exhausting the regenerative ability of midstorey individuals potentially leads to a site that has few if any trees in the middle size classes, with little chance of successful recruitment from seed germination. This is essentially the type of community documented at Site 17 (see Chapter 3) where an annual fire regime has been in place for some years as a result of burning practices conducted by the local landholder. These types of community fit the "interruption of maturation and developmental growth" classification for plant population decline provided by Keith (1996) (Table 5.1). Keith (1996) described how high frequency fires not only cause the death of pre-reproductive juvenile plants (particularly affecting obligate seeders), but also cause the death of pre-resistant juvenile plants (affecting resprouters). Even though *M. viridiflora* lies somewhere along the continuum between obligate seeder and resprouter (a facultative resprouter, Keeley 1986), it seems unlikely that as a species it is well adapted to continually high fire frequencies.

The seedling recruitment investigation also supports this argument (Figure 5.4). Less than three years after fire, recruitment had almost returned to pre-fire levels. Elevated levels of seedling density following fire were maintained longer at Site 10, a site with a gradational soil profile and high surface soil moisture levels (Chapter 4). Site 5 has a soil type similar to Site 8 (duplex soil with lower surface moisture levels) which may explain the poorer success of fire-induced seedlings. Alternatively, other site factors such as variable predation rates and competition for resources may also be important.

Irrespective of the controlling environmental variables, it appears that repeated fires (particularly annual fires) would prevent most juveniles establishing to maturity. At the end of the study period, the surviving seedlings were still less than 30 cm tall, indicating they would be unlikely to have developed sufficient bud or lignotuber reserves to withstand a further fire. Furthermore, an investigation of the seed bank at Site 5 (including application of a heat treatment to the soil) produced no germination of *M. viridiflora* (Skull 1991). Bradstock (1990) highlighted the importance of the seed bank to facultative resprouters in terms of their long-term longevity at frequently fired sites. A lack of available seed bank resources further suggests that frequent fires will lead to the decline of these Tea-tree communities.

The annual fire regime at Site 17 was halted by the now QDE for the first time in 1996. As a result the site provides a unique opportunity to monitor the possible "recovery" of community structure as recruiting individuals have the potential to grow beyond the understorey. It may be some years before this can occur given that seed-banks are likely to be depleted, as is the regenerative ability of many potential recruits. Alternatively, the observed community structure may be independent of fire. This suggestion has been advanced by some researchers working on other (eucalypt-dominated) tropical plant communities, with structural diversity not necessarily being restored in the absence of fire. Whatever the final outcome, the site certainly provides an excellent avenue for future research. A long-term study would allow various aspects of the response of this community to further fire and fire exclusion to be modelled, and subsequently compared with models developed for plant communities elsewhere in Australia (e.g. Whelan 1995).

The simulated fire treatment employed in this experiment is not without its limitations. It is extremely unlikely that any treatment of this nature could replicate a "natural" fire in terms of real fire residence times, fire temperature profiles, fire intensity variability as a result of spatial and temporal variations in fuel loads, wind speed, topography and so on. This problem is, however, not unique to this investigation. The problem of adequately simulating fires has been lamented by many researchers (e.g. Hobbs *et al.* 1984) who have concluded that there may well be few, if any, practical solutions to these problems.

Other plant species in the understorey of these woodlands are also affected by fire. Grasstrees (*X. johnsonii*) exhibited a massive increase in density following fire, a finding reported by other researchers (Bulow-Olsen *et al.* 1982). This increase was sustained for nearly two years after the fire event (Figure 5.3). It is debatable whether this effect will be sustained in the long-term, with many of the new seedlings recorded growing adjacent to well established clumps of individuals. Intraspecific competition may prevent the successful establishment of these juveniles through to maturity. Given the slow growth of this species documented elsewhere (Bulow-Olsen *et al.* 1982), it could be some time before questions such as this could be investigated and possibly answered through further research.

The maximum fuel loads documented at Sites 5 and 6 before fire (5.2 and 8.8 t ha^{-1} respectively) fall towards the lower end of the spectrum recorded for highly productive floodplain *Melaleuca* forests in the Northern Territory (Finlayson *et al.* 1993). The Site 6 values are, however, close to those calculated for tropical rainforests (Spain 1984). At other sites investigated in Chapter 3 (particularly Site 7), although no samples were actually collected, field records indicate fuel loads that were nearly waist-deep in places (tall grasses and dense *X. johnsonii*), and therefore likely to be well over 10 t ha^{-1}. This is probably related to the fact that Site 7 is isolated (surrounded by closed forest) and may, therefore, actually have even lower fire frequencies than initially estimated. Even though the fuel loads documented here for *M. viridiflora* sites are below those recorded for most tall eucalypt forest communities across Australia (described in more detail below), they compare with many of the tall open grassy woodland communities (5-10 t ha^{-1}) described by Gill *et al.* (1991) in northern Australia. Thus the contribution of organic matter turnover by these communities to general ecosystem nutrient cycling and detrital food chains is probably significant.

The maximum (pre-fire) fuel loads recorded at the tall eucalypt forest site (Site E, 10-20 t ha^{-1}) are comparable to those found in other studies in similar forest types around Australia (Raison *et al.* 1983; O'Connell 1987; Fensham 1992). The only higher records (up to 30 t ha^{-1}) are for the Karri (*Eucalyptus diversicolor* F. Muell.) tall open forests of south-western Australia (O'Connell 1987). As with the *M. viridiflora* woodlands, the recovery of the fuel loads to pre-fire levels at the *E. cloeziana* site appears to be a slow process. Almost two years after fire, the fuel load at Site E was less than 50% of the pre-fire level. The time required to reach pre-fire levels may be several years. As a result, fuel reduction burns need not be conducted as often as once thought by management agencies (QDEH 1996a).

The fuel load patterns at Site 5 also appear similar to those documented elsewhere (Walker 1981), with fuel load peaks occurring in summer once the initial post-fire growth phase has levelled out. Recovery of the fuel loads present in the understoreys of the *M. viridiflora* communities studied is also slow. Total fuel loads were still only 70% of the pre-fire level nearly three years after fire at Site 5 following an early rapid rise (Figure 5.6). Although the data for Site 6 (Figure 5.7) showed a similar early rapid rise, it may also be that this does not reach pre-fire levels in the short-term. There are many factors affecting fuel load accumulation (Fensham 1992), any number or combination of which could explain the variations in patterns observed. These include subtle changes in slope, patchiness of any previous burn(s), differences in soil fertility, patchiness inherent in the plant communities themselves and different rainfall patterns.

It is of course possible that the pre-fire fuel loads recorded in this study were not entirely representative of general fuel load patterns in the absence of disturbance. Sufficient time did not permit long-term sampling prior to the fire events which would have allowed the scale of natural temporal fluctuations in the fuel loads of the communities to be determined. The fact that accumulation is slow, however, seems to indicate that low fire frequencies would be preferable to relatively high fire frequencies (i.e. fire no more frequent than every three to five years).

Moderate to high fire frequencies are currently accepted as suitable for a range of plant communities in north-eastern Queensland, even though they may not actually be appropriate from an ecological viewpoint. The inappropriateness of high fire frequencies has already been shown to be the case for other vegetation types that have been subject to detailed research elsewhere in Australia (e.g. Fensham 1992; Williams & Gill 1995) and overseas (e.g. Robbins & Myers 1992). This is particularly the case when fires are continually ignited in the same season (low intensity) for fuel reduction purposes, potentially leading to localised plant extinctions, introductions of exotic species, alterations to seed germination responses and a reduction in habitat quality for fauna (Robbins & Myers 1992). Thus there is a need for management agencies to avoid being overly prescriptive with fire regimes, especially when adequate data is unavailable.

Unfortunately, little is known about the effect of fire on many vegetation types throughout Australia (Gill *et al.* 1991), in particular those of northern Australia (Gill *et al.* 1996). The WTBR of Queensland is certainly no exception, with management agencies generally adopting and refining "best-guess" approaches (Mr P. Stanton, pers. comm.). This lack of research is the result of inadequate funding, a lack of recognition by governments that fire is widely used as a key management tool in conservation reserves and grazing lands, and a similar lack of recognition that fire remains one of the most important natural ecological disturbance agents present in the Australian landscape.

Relatively high fire frequencies certainly do not appear to be appropriate in the case of remnant *M. viridiflora* woodland management. These woodlands, unlike so many of the open eucalypt communities within the WTBR, are not threatened in terms of habitat conversion by the process of rainforest invasion. *Melaleuca viridiflora* woodlands usually occupy areas with poorly drained soil, and the establishment and succession of other communities in this seasonally inundated, specialist environment is highly unlikely. This notion is supported by an analysis of aerial photographs that show, in areas where this community has remained intact over the past 50 years, patch size and boundary location (with adjoining plant communities) have remained essentially unchanged (see Chapter 2).

Current QDE fire management recommendations for *M. viridiflora* communities range from one utilising fire every two to three years (based on recommendations for orchid conservation, Lavarack 1994) to one fire every ten years as a result of the work conducted during this study (QDEH 1995b). Fires every two or three years may be suitable for very large patches of this community where a mosaic of patches with different fire frequencies could be created. This mosaic is not likely to be the end result of such good intentions, however, with most remnant patches (especially on the mainland) being quite small (see Chapter 2). Additionally, for many large patches of remnant habitat there is little chance of controlling prescribed fires of this nature without relatively high levels of disturbance (fire break creation) occurring. Furthermore, on the mainland many unplanned fires are recorded every year along the Bruce Highway where many remnants of these communities occur. These fires are generated by careless members of the general public, arsonists and poor land management practices.

Recommendations to management agencies resulting from this research (for reduced fire frequencies) are based upon the comparative structural integrity of sites that have been burned on a ten yearly basis in recent times. These include Site 5 documented in this study (Chapter 3), and those studied on other tropical floodplains with even lower fire frequencies (Dr M. Finlayson, pers. comm.). The recommendations are also based upon fuel load recovery curves and the need to adopt a precautionary approach until more research information comes to hand.

Significantly, the effects of fire on the fauna associated with this plant community have not been investigated. It needs to be recognised that recommendations with respect to the management of any habitat should not be formulated on the basis of vegetation assessment alone. Fauna studies are required to provide the additional information required by management agencies. The availability of this information could promote the likelihood of management decisions being ecologically sustainable. Other factors that should be considered when developing a fire management plan for an area (including fire intervals) are discussed by Williams and Gill (1995). Furthermore, numerous authors have outlined the "minimal data sets" required to adequately monitor the effects of fire in conservation reserves (e.g. Gill & Nichols 1989; Good 1989; Wardell-Johnson *et al.* 1989).

It is also important that the recommendations formulated from this and other research programs are not prescriptively applied outside the region in which they were generated. Whilst the sampling units have been replicated, the main experimental factor examined here (repeated prescribed fire) was not. To replicate the fires was, as I have previously indicated, impractical (hence the use of the simulated fire experiment) and, given the areas burnt and the random sampling process used, perhaps unnecessary (Dr B. Loneragan, pers. comm.). The problem of extrapolating results to other regions has been raised previously by both conservation managers and researchers (e.g Stanton 1992; Williams *et al.* 1994), and an example comes from recent research conducted on Cape

York Peninsula (Crowley & Garnett 1995).

This research concludes that extended fire-free fire intervals in *M. viridiflora* woodland communities (similar to those recommended here) are the primary cause of significant habitat loss for the endangered Golden-shouldered parrot. The parrot inhabits grasslands which are being overgrown by *M. viridiflora* (eventually converting the grassland to woodland) which is usually kept in the understorey as a result of relatively frequent firing practices. Differences in natural and imposed fire frequencies, land use histories and other ecological properties of ecosystems (the differences in climate between such regions have been demonstrated and discussed in Chapter 3) necessitate the adoption of potentially very different management approaches, even in adjacent regions. Great care must be exercised to ensure that the best management intentions of today do not become the management problems of tomorrow.