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CHAPTER SIX - EXOTIC SPECIES INVASION INTO

MELALEUCA VIRIDIFLORA OPEN WOODLANDS:

PINUS CARIBAEA MORELET VAR. *HONDURENSIS* BARRETT & GOLFARI (PCH)

- A CASE STUDY

*Conservation biology differs from most other biological sciences
in one important way: it is often a crisis discipline* (Soulé 1990).

6.1   INTRODUCTION

6.1.1   Pine invasion

Invasions of remnant terrestrial habitats by exotic species is one of the major threats to biodiversity worldwide (Hobbs & Humphries 1995). Invasions have caused the local and regional extinction of some plant species, and in extreme cases have contributed to their complete extinction (Adair 1995). *Pinus* species (including *P. caribaea*) are already considered to pose a serious threat to natural habitats in Australia (Humphries *et al.* 1991), and have been shown to cause reductions in species richness following invasion into South African fynbos (Richardson *et al.* 1989).

Cronk and Fuller (1995) listed the following characteristics typical of species capable of invading open woodland habitats; high levels of seed production, seed dispersed by wind and/or birds, rapid growth rates, early reproductive maturity and a degree of shade tolerance. Research on temperate species of invasive pines (particularly *Pinus radiata* D. Don) has shown that pines have all of these characteristics (Burdon & Chilvers 1994). Fox (1995) added that plant species could either alter available resource levels (e.g. nitrogen fixers) or the existing disturbance regime to be effective invaders. The concept of invading species altering ecosystems is not new, with Wells *et al.* (1986) defining a transformer species as one that "changes the character, condition, form or nature of a natural ecosystem over a substantial area". Pine species are capable of changing several ecological characteristics of ecosystems, with those documented in temperate regions of

Australia discussed in more detail below.

Following a comprehensive review of pine invasions worldwide, Richardson and Bond (1991) added several further reasons to explain why pines were so successful at invading ecosystems outside their natural range. These included pines being drought tolerant, possessing the ability to survive on soils of low fertility, and individual, isolated invaders being able to self-pollinate and therefore give rise to new colonies. In a review of the threat of environmental weeds to biodiversity in Australia, Adair (1995) listed 12 characters of plants indicating a potential for weediness in native ecosystems. Nine of these characters apply to pines with a further two remaining possibilities subject to the results of future research.

A great deal of research has focussed on gaining an improved understanding of the processes controlling pine distribution in a variety of habitats (e.g. Richardson & Bond 1991; Hobbs & Huenneke 1992; Myster 1993). The importance of a "window of invasion" where favourable biotic and abiotic conditions exist for dispersal, germination and seedling establishment has been highlighted by several researchers (e.g. Johnstone 1986; Myster 1993). The availability of light, for example, has been shown to be particularly important for pine seedling establishment, with open habitats (grasslands, shrublands and open woodlands) the most affected (Richardson & Bond 1991). Moreover, invasion by pines is most likely to occur in habitats where competition in the regeneration niche (part of a plant's total niche as originally defined by Grubb 1977) is low i.e. few or no other species with a similar growth form are present (Richardson & Bond 1991). It is also acknowledged that many interacting ecological factors are often important, particularly the regime of ecosystem disturbances including fire, grazing, soil disturbances, altered nutrient inputs, trampling and habitat fragmentation (Hobbs & Huenneke 1992).

Second to the actual process of clearing and planting associated with plantation establishment {exogenous ecological disturbances as defined by Fox & Fox (1986b)}, the effects of fire on pine invasion are the most relevant to this investigation. Fire is usually an endogenous disturbance in ecosystems, and the challenge for management agencies is

to use fire regimes that control or reduce the distribution of exotic species, whilst providing a competitive advantage for native species (Adair 1995). Cronk and Fuller (1995) indicated that *P. radiata* was well adapted to fire as the seeds are protected in a serotinous cone, and fire stimulates the release of seed and can also create favourable conditions for germination. Fire and fire suppression have both been shown to facilitate pine invasion elsewhere (Richardson & Bond 1991). The latter can result in grasses becoming moribund, thus reducing competition for similar resources (Richardson & Bond 1991). The effects of fire are, however, species-specific and depend upon the fire regime (season, intensity and frequency) and species life history traits (Hobbs & Huenneke 1992; Keith 1996).

No previous research anywhere in the world has documented any variety of *P. caribaea* invading remnant plant communities (Dr D. Richardson, pers. comm.). The pine species studied here (the Caribbean pine, Pinus *caribaea* var. *hondurensis*) is, however, an important timber species in north-eastern Queensland and, as a result, much research has been conducted on genetic improvement, growth rates in plantations, and germination and seedling establishment under controlled laboratory conditions (e.g. Lamb 1973; Woolaston *et al.* 1990). Unpublished research conducted by the Queensland Forest Research Institute has shown that this species:

·   flowers around May each year and releases seed soon thereafter;
·   has seed that is viable for only short periods of time once released from the seed cone;
·   has seed which is killed by fire and will not germinate in the absence of soil moisture (although no field data are available). Similar findings of reduced levels of germination at low soil moistures were recorded as a result of research on the germination of *P. radiata* during a drought (Chilvers & Burdon 1983);
·   can be killed by moderate intensity fires if individuals are still relatively young (Dr M. Dieters, pers. comm.); and
·   has recorded diameter increases of more than 17 cm month^{-1} in north-eastern Queensland plantations (Woolaston *et al.* 1990).

The most detailed and long-term study of pine invasion in Australia has been conducted by researchers in the Australian Capital Territory, examining the pattern and processes associated with *P. radiata* invading temperate eucalypt forest (Burdon & Chilvers 1977; Chilvers & Burdon 1983; Burdon & Chilvers 1994). The length of this study has enabled the following important ecological points to be documented:

- pines had displaced eucalypts despite high mortality rates of pine seedlings;
- pines had much higher growth rates than the eucalypts;
- the growth of eucalypt seedlings were completely suppressed by high shade conditions under larger pine trees; and
- alterations to significant community characteristics including fire regimes (higher flammability as a result of blankets of pine needles on the forest floor), increased shade levels and the appearance of mycorrhizal fungi usually only associated with pines.

The authors of this research concluded that the decline in the eucalypts which they documented was expected to continue, particularly as a result of the development of competition between the two species. This has serious implications for the long-term viability of the eucalypt community and its conservation value (without management intervention).

The management of biological invasions has been the focus of relatively recent reviews in Australia (Humphries *et al.* 1991; Hobbs & Huenneke 1992; Adair 1995; Fox 1995; Hobbs & Humphries 1995). Most current theory suggests that management practices should move away from simply targeting the species of interest and focus instead on the ecosystem as a whole in conjunction with relevant socio-economic factors. The ability of invading species to alter ecosystem functioning implies that land managers need to be willing to alter management practices accordingly to ensure conservation objectives can still be achieved.

The importance of preventing invasions or, failing that, the early detection of and rapid response (assessment, control, rehabilitation) to invasions are common topics addressed in

the literature (e.g. Adair 1995; Fox 1995; Panetta & Scott 1995). Costs of invasion management are significantly increased when invasions are well advanced and, by this stage, ecosystem conservation values have usually been reduced (Hobbs & Humphries 1995). Manual removal of exotic species is considered to be the most environmentally sensitive technique (Bradley 1988) and, although little recognised by many management agencies, has contributed much to the conservation of plant communities in Australia (Adair 1995). The technique is, however, claimed to be labour-intensive and therefore expensive where volunteers are not available (Hobbs & Humphries 1995). In addition, if large numbers of individual plants need to be removed, the soil disturbance associated with removal may facilitate further invasion (Richardson & Bond 1991). Successful removal of juvenile pines by hand without any other necessary treatment has been reported for *Pinus pinaster* Aiton, although a combination of cutting followed by burning is often employed (Cronk & Fuller 1995).

Many other management techniques also exist, including biological, chemical, reproductive and breeding control strategies. No attempt is made to review these here and the reader is referred to Fox (1995) for further information. It should, however, be noted that successful control usually involves a range of techniques including ecosystem disturbance management (e.g. fire), a form of "ecological control" which was first defined by Groves (1989). The use of many techniques in an integrated fashion has achieved best success in agricultural landscapes, with similar attempts in conservation reserves less fruitful, primarily because fewer alternatives are available and/or acceptable for use (Adair 1995).

Current DNR recommendations for the control of pines in tropical lowland communities dictate the use of herbicides for trees less than 1.5 m tall and a cut stump injection technique for larger individuals (Mr C. Dean, pers. comm.). The current management philosophy is to only manage pine invasions where a significant land tenure is threatened, and to date this has only been carried out once in Queensland, at Cooloola National Park (Mr K. Gould, pers. comm.).

6.1.2 Germination experiments

The effective management of plant species for conservation purposes requires a knowledge of the processes affecting population dynamics (Leigh *et al.* 1984). The critical role of germination from the soil seed bank in plant population dynamics has been reviewed by several authors (e.g. Garwood 1989), with the effects of fire potentially having great importance (Whelan 1995). Lamont *et al.* (1991) showed that both fire and soil moisture were crucial determinants of seed release, seed germination, seedling establishment, and hence recruitment of some plant species in fire-prone habitats.

In a detailed review of the germination of native Australian plant species, Langkamp (1987) indicated that the optimum germination temperature for *M. viridiflora* was 30°C. Turnbull (1986) noted that seeds will germinate readily without any treatment at all, provided adequate light is available. This species flowers mainly between April and August, although trees in wetter areas have flowered as late as December (Skull 1995). Fruits are usually mature in spring and early summer (July-December) (Turnbull 1986).

Lamb (1973) provides a comprehensive assessment of information from all over the globe regarding the growth of *P. caribaea* var. *hondurensis* in plantations. This report does not list specific germination requirements of the species, although it does indicate that all varieties of *P. caribaea* germinate readily under tropical conditions. Soaking seed in water under laboratory conditions has produced higher rates of germination, indicating a potential tolerance to at least moderate levels of soil moisture in the field (Lamb 1973). To date no data are available regarding germination of this species under field conditions (Dr M. Dieters, pers. comm.). In plantations cones usually release seed between May and June, with reproductive maturity usually reached 5 to 10 years after the date of planting (Lamb 1973). It is important to note, however, that as plantations are fertilised and managed to reduce exotic species and "overcrowding", growth rates and maturation times may be faster than those of individual trees invading other habitats (Dr M. Dieters, pers. comm).

6.1.3   Aims of this investigation

The specific objectives of this section of the thesis were to:

(i)   examine the patterns of Caribbean pine (*P. caribaea* var. *hondurensis*) invasion from an adjacent plantation into a remnant *M. viridiflora* woodland;

(ii)   determine if the distribution of *P. caribaea* var. *hondurensis* in the woodland relates to that of any of the dominant native species;

(iii)   compare the growth rates of *M. viridiflora* and *P. caribaea* var. *hondurensis*;

(iv)   determine the effectiveness of prescribed burning as a control measure for *P. caribaea* var. *hondurensis*; and

(v)   examine the effects of both soil temperature and soil moisture on the germination of these species.

## 6.2   METHODS

6.2.1   Pine invasion

The majority of this study was conducted on private property near Cardwell, adjacent to Caribbean pine plantations planted in 1978 by the Queensland State Forest Service (QSFS). Three parallel, 400 m long transects 20 m apart were established in a remnant *M. viridiflora* open woodland community (*M. viridiflora* study Site 14 from Chapter 3 was located approximately in the middle of the transects as it was established prior to this investigation-throughout this chapter it is referred to as Site 3), surrounded by either lowland closed forest (dominated by *Lophostemon suaveolens*, *Acacia mangium*, *Alphitonia petriei* C.T. White & Braid and *Corymbia clarksoniana*), plantation pine, fire breaks or the Cardwell Forest Drive (Figure 6.1). The transects were positioned on a bearing of 345° which enabled the maximum distance through the woodland to be sampled. In addition, this bearing approximates that of the south-easterly airstream which usually prevails when the pines are known to be releasing seed (Bureau of Meteorology data; Dr M. Dieters, pers. comm.; Lamb 1973). Fire had been excluded from this area for approximately ten years (Mr P. Pomroy, pers. comm.).

Figure 6.1   Map showing location of *P. caribaea* var. *hondurensis* invasion study sites.
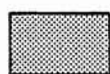
3 parallel transects

CARDWELL →

ELLERBECK ROAD

firebreak

3

1

2

5

4

6

7

8

Cardwell
Forest Drive

firebreak

N
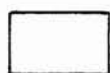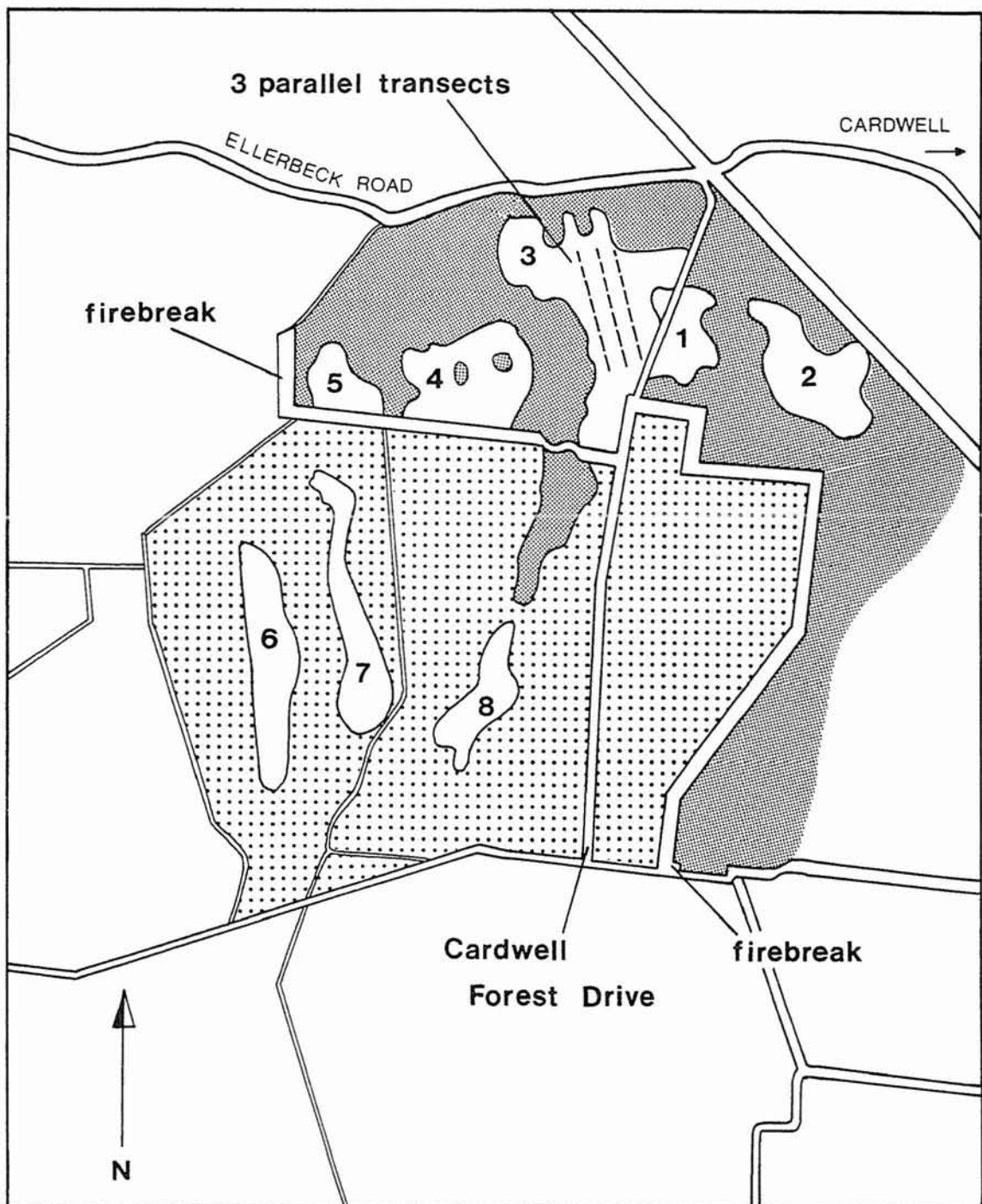
Melaleuca viridiflora woodland
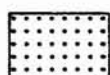
Pinus caribaea var. *hondurensis* plantation

Lowland closed forest

The three transects were divided into forty contiguous 10 m x 10 m quadrats. In each quadrat the following information was recorded:

the number of individuals in two size classes (<1.5 m and >1.5 m) was recorded for the following dominant species; *M. viridiflora*, *P. caribaea* var. *hondurensis*, *Acacia crassicarpa*, *Acacia mangium*, *Casuarina equisetifolia* var. *incana*, *Grevillea pteridifolia*, and *Lophostemon suaveolens*. These size classes were chosen as previous work on pines suggests that when trees become taller than 1.5 m, control initiatives other than fire are required. Furthermore, provided mature trees are recounted, any remaining changes in the >1.5 m size class can be attributed to the sapling (midstorey) category; and

the number of *X. johnsonii* present in the understorey.

The DBH and height of all pine trees recorded from Transect 2 were also documented.

To highlight the ability of *P. caribaea* var. *hondurensis* to invade remnant *M. viridiflora*-dominated woodlands, a further eight sites within the Cardwell State Forest were investigated. Five of these sites (Sites 1-5) were located just to the north of the State Forest (all less than 500 m from the nearest plantation), whereas three sites (Sites 6-8) were completely surrounded by plantation pine (Figure 6.1). These sites (except for Site 3) were too small to allow transects to be sampled, so an alternative sampling procedure was adopted.

At each of the eight sites, 5 well-spaced (approximately 10-20 m) plots (10 x 10 m) were established along a transect, similar to the manner in which sites were established in Chapter 3 (the relevant data for Site 3 has been taken directly from Chapter 3-Site 14). Within these plots the density, height and DBH of all *P. caribaea* individuals, and the general structure of the woodland community were recorded (Site 3 was necessarily re-assessed for pine trees). To assess the effectiveness of fire as a management tool for the control of *P. caribaea* var. *hondurensis*, a prescribed fire was ignited at the transect site in October 1995 (for full details see Appendix E). Note that although *X. johnsonii* was not recorded at Site 14 in Chapter 3, it was documented (and therefore selected for analysis)

in the transects during this investigation.

Finally, the field growth rates of *P. caribaea* var. *hondurensis* and *M. viridiflora* were compared. Thirty individuals of each species growing at Site 3 were tagged and both DBH and height measurements recorded in August 1993, December 1993, February 1994, May 1994, August 1994 and June 1995.

6.2.2 Germination experiments

All procedures used in the germination experiment followed the standards and methods recommended by the International Seed Testing Association (ISTA 1985). Seeds of *M. viridiflora* were collected from trees at Site 3, whereas those of *P. caribaea* var. *hondurensis* were obtained from the QSFS. From the temperature trial experiment (Section 5.3.1), the following heat treatments were selected as suitable for investigation in the germination experiment: 30°C, 60°C, 100°C, 150°C, 200°C, and 600°C. This temperature range also covered those recommended as likely to be reached during fire in this vegetation type (Mr K. Tolhurst, pers. comm.). Thirty degrees Celsius was also assessed as this has previously been documented as the optimal germination temperature for *M. viridiflora* (Langkamp 1987). Additionally, this treatment acted as a control to ensure that the seed had adequate viability.

Seeds in treatments up to and including 100°C were heated for approximately 30 seconds in a calibrated conventional oven. For the treatments above 100°C, seeds were heated for three seconds in a crucible using a muffle furnace. Temperature durations longer than these will only result in seed death, and are longer than usual fire residence times (Dr M. Gill, pers. comm.).

Melaleuca viridiflora has very small seeds that are difficult to separate from the general seed chaff. Peterson (1985) recommended that for this type of seed, weighed replicates of the seed/chaff mix should be used in preference to seed counts. Ten replicates of 0.05 g were weighed and transferred to a nine cm diameter petri dish that had been pre-lined with two moistened filter papers. Seed chaff was spread evenly to prevent fungal

establishment.

Thirty *P. caribaea* var. *hondurensis* seeds were counted into each of another ten, similarly prepared petri dishes.This experiment was conducted in a 30°C constant temperature room with a 12 hour light/12 hour dark cycle. Germination was defined as the stage at which the emerging radicle extended to twice the length of the seed (Lyons 1989). Seeds were monitored for a total of 28 days. Petri-dishes were kept moist with de-ionised water applied twice daily. De-ionised water was used to further reduce the level of contaminants affecting germinating seeds. *Fungarid* fungicide was applied to the petri-dishes (according to the manufacturers instructions) once every five days. Aseptic techniques were used at every stage of the experiment to minimise fungal attack.

To determine the effect of soil moisture on the germination of *M. viridiflora* and *P. caribaea* var. *hondurensis*, six moisture levels were examined: 0%, 10%, 20%, 25%, 30% (approximate field capacity) and 40%. This range of values encompassed those documented at the three *M. viridiflora* sites investigated in Chapter 4 (Figure 4.2). Seeds of both species were weighed or counted as described above. To maintain accurate soil moisture levels, no filter paper was used in this experiment, as the absorption of water by filter paper is not linear (Mr D. Pollock, pers. comm.). Instead, sterilised sand (steamed at 120°C for 15 minutes) was used following sifting through a one mm sieve. Four hundred grams of this soil was then spread evenly to a depth of 0.5 cm in the bottom of each petri dish. Ten replicates of each soil moisture level were prepared. Water was added volumetrically to simulate each soil moisture level. The petri dishes were then sealed with masking tape so that each soil moisture level was maintained throughout the life of the experiment (28 days).

6.2.3 Data analysis

To determine if any of the seven species assessed on the three transects had any significant effect on the distribution of *P. caribaea* var. *hondurensis*, multiple linear regressions using backward stepwise variable elimination were conducted. Data for the three transects were pooled, and total pine density (both size classes pooled), all pines greater than 1.5 m and all pines less than 1.5 m were analysed. Backward stepwise regression procedures initially analyse all variables (species) in the data set simultaneously. The procedure then progressively eliminates variables with the least

significant change at each step of the analysis, until critical numerical limits are reached. Thus, only the variables responsible for most of the variance remain in the regression equation.

A repeated measures ANOVA was used to examine the growth (DBH and height) data collected for *P. caribaea* var. *hondurensis* and *M. viridiflora* at the pine invasion study site (Site 3). The effects of fire on the total density (all size classes and transects pooled) of all species was tested using a multi-variate ANOVA. A two-way ANOVA was used to assess the germination responses of the two species of interest to different temperatures and moisture levels. SAS Version 6.0 was used for the repeated measure ANOVAs and SPPS Version 6.0 for all other data analysis procedures.

6.3   RESULTS

6.3.1   Pine invasion

A summary of the results from the combined transects and eight other sites investigated within the State Forest are presented in Table 6.1. There is no readily apparent relationship between the density of invasion and the location of each site with respect to the pine plantations. Although Site 7 recorded the highest number of pines, some sites to the north of the State Forest (particularly Sites 3 and 4) had similar or higher densities than those completely surrounded by pine. Four of the sites investigated (Sites 1, 6, 7 and 8) only had pines in the less than 1.5 m size class, suggesting that either invasion is relatively recent or growth rates are particularly slow. Invasion densities were low at sites where the densities of either *M. viridiflora* or *X. johnsonii* were high (Sites 5 and 1 respectively). No trend in the number of pines (or their sizes) was observed with respect to the total density of the six other species recorded. Pine trees establishing at the transect site are pictured in Plates 3a-b. These photographs (all taken on the same date) indicate that several "windows of invasion" have been exploited by the pines, as seedlings (Plate 3a) and juveniles (Plate 3b) co-exist at the site. Additionally, individuals well over 4 m tall occur at the site.

Table 6.1   **Density and size of *P. caribaea* var. *hondurensis* (P.c.) in eight *M. viridiflora* (M.v.) woodland communities.** Pooled data from the three transects are also given. Means and standard errors are listed in parentheses for the size ranges of P.c. and the three transects. All figures are for 500 m^{-2}. An asterisk (*) indicates the site is completely surrounded by plantation pine.

Site	No. P.c. total	No. P.c. >1.5 m	No. P.c. <1.5 m	DBH range P.c. (mm)	Height range P.c. (cm)	No. M.v. >1.5	No. M.v. <1.5	Total no. of all other dominants	No. X.j.
1	9	0	9	3.9-22.6 (13.8±2.7)	43-110 (82.9±7.9)	43	75	13	242
2	5	2	3	10.9-56.5 (31.7±9.9)	77-410 (219.3±74.3)	62	50	15	0
3	13	1	12	8.5-39.8 (19.6±2.4)	61-178 (95.2±9.3)	83	123	92	0
4	27	5	22	2.1-38.7 (13.9±1.8)	27-270 (99.2±10.6)	74	53	45	130
5	9	3	6	3.6-49.6 (20.6±4.6)	50.5-370 (154.5±37.6)	273	523	19	6
6*	12	0	12	1.5-52.0 (7.23±4.1)	18-84 (43.0±5.4)	89	58	15	20
7*	78	0	78	0.8-21.2 (7.5±0.6)	26-138 (78.8±3.3)	37	79	107	3
8*	13	0	13	0.9-15.5 (3.8±0.3)	11-220 (47.7±3.3)	171	170	49	0
Transects	8	6	2	2.0-52.7 (16.8±1.4)	44-412 (164±11.0)	114	259	100	10

Plate 3 Caribbean pine (*P. caribaea* var. *hondurensis*) invasion into *M. viridiflora* open woodland habitat (Site 14 near Cardwell).

(a)   *P. caribaea* var. *hondurensis* seedling.

(b)   High level of *P. caribaea* var. *hondurensis* invasion.

The density of *M. viridiflora*, *P. caribaea* var. *hondurensis* and *Acacia mangium* (the dominant closed forest species) along each of the transects is presented in Figures 6.2, 6.3 and 6.4 respectively. The density of *M. viridiflora* in both size classes exhibited similar trends along each of the transects (Figures 6.2a-b). Densities were low at two locations (0-50 m and around 180 m), with noticeable rises in the 80 to 100 m section and towards the end of the transect. Pine trees in both size classes were abundant at the margin of the woodland (0 m), with only the larger individuals recording a pronounced peak between 80 and 180 m (Figures 6.3a-b). Densities along the remainder of each transect (200-400 m) were low. *Acacia mangium*, although having several small peaks along the transects in both size classes, was markedly more abundant towards the end of the transects where *M. viridiflora* woodland graded into closed forest (Figures 6.4a-b).

The mean DBH and height of *P. caribaea* var. *hondurensis* peaked in the 80 to 180 m region of the transect (Figure 6.5a), as did the range of heights (Figure 6.5b). The DBH data points showed nearly identical trends to the height data and have therefore not been presented.

Results of the multiple regression of the pooled species data indicated a weak positive correlation between the distribution of pine trees and three other species (Table 6.2). *Acacia mangium* had a negative relationship (increasing towards the closed forest which it dominated, Figure 6.4), whilst *A. crassicarpa* and *C. equisetifolia* var. *incana* had slightly positive correlations. When the data were analysed using the two size classes separately, pine trees less than 1.5 m exhibited very weak correlations with *A. mangium* (negative) and *A. crassicarpa* (positive) (Table 6.3). In contrast, a much stronger relationship was evident between pine trees more than 1.5 m and both size classes of *C. equisetifolia* var. *incana* (Table 6.4). In all of these analyses, *X. johnsonii* was the first species eliminated from each of the regression procedures.

Figure 6.2   Distribution of *M. viridiflora* densities (100 m^{-2}) on each of the three transects at Site 3.

     (a)   Individuals > 1.5 m

     (b)   Individuals < 1.5 m

(a)

(b)

Density 100 m^{-2}

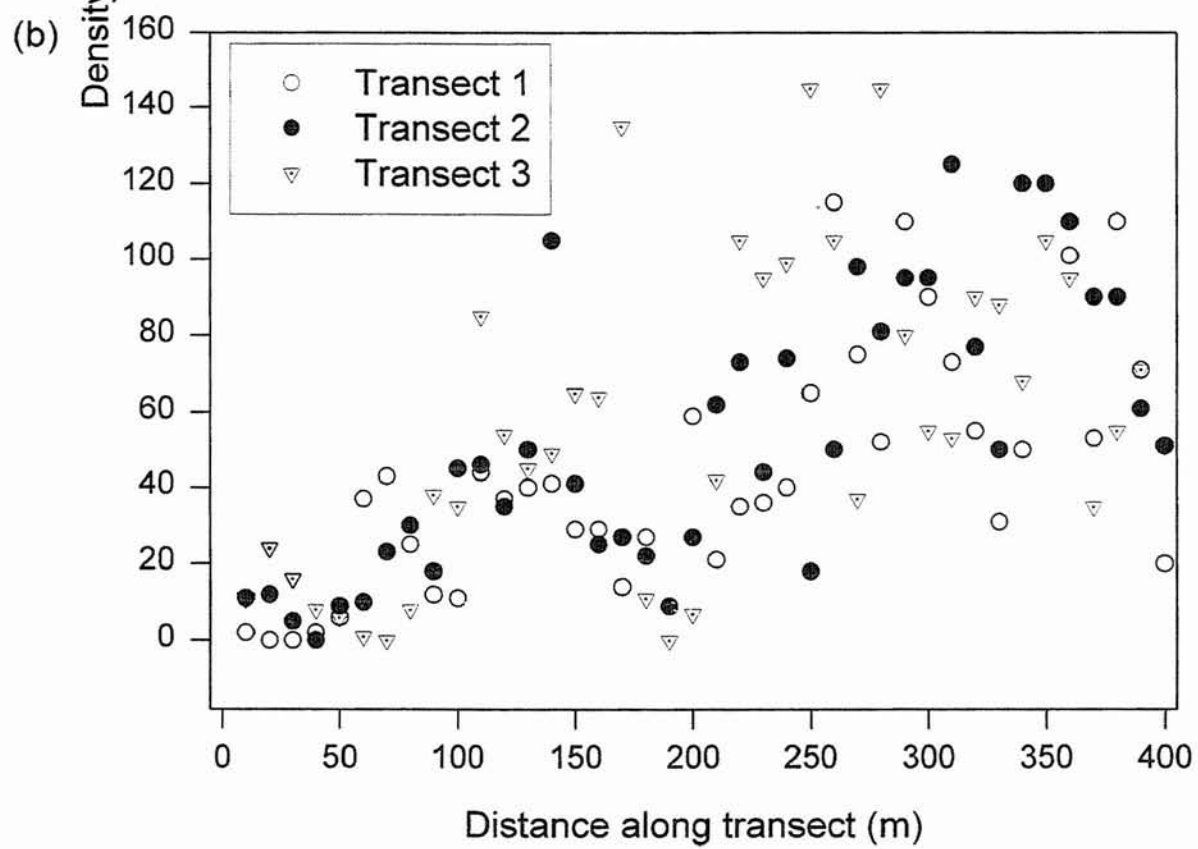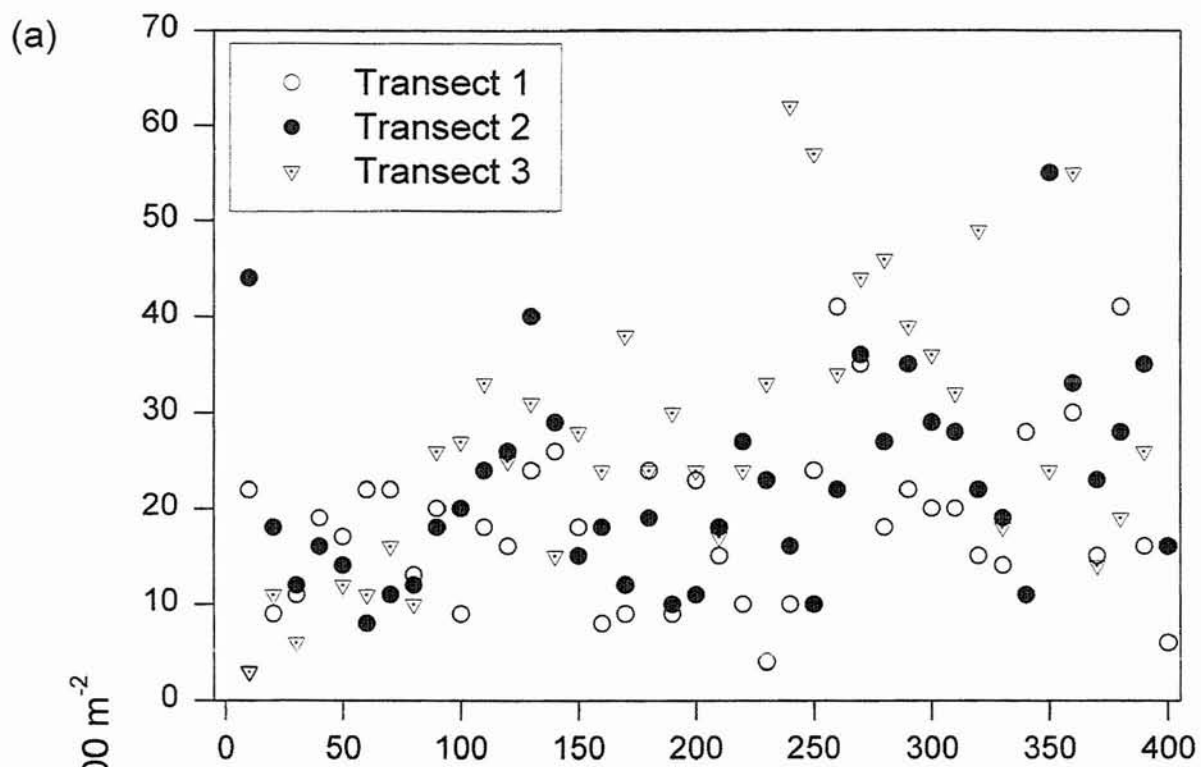Distance along transect (m)

Figure 6.3   Distribution of *P. caribaea* var. *hondurensis* densities (100 m^{-2}) on each
                of the three transects at Site 3.

                (a)   Individuals > 1.5 m

                (b)   Individuals < 1.5 m

(a)

(b)

Distance along transect (m)

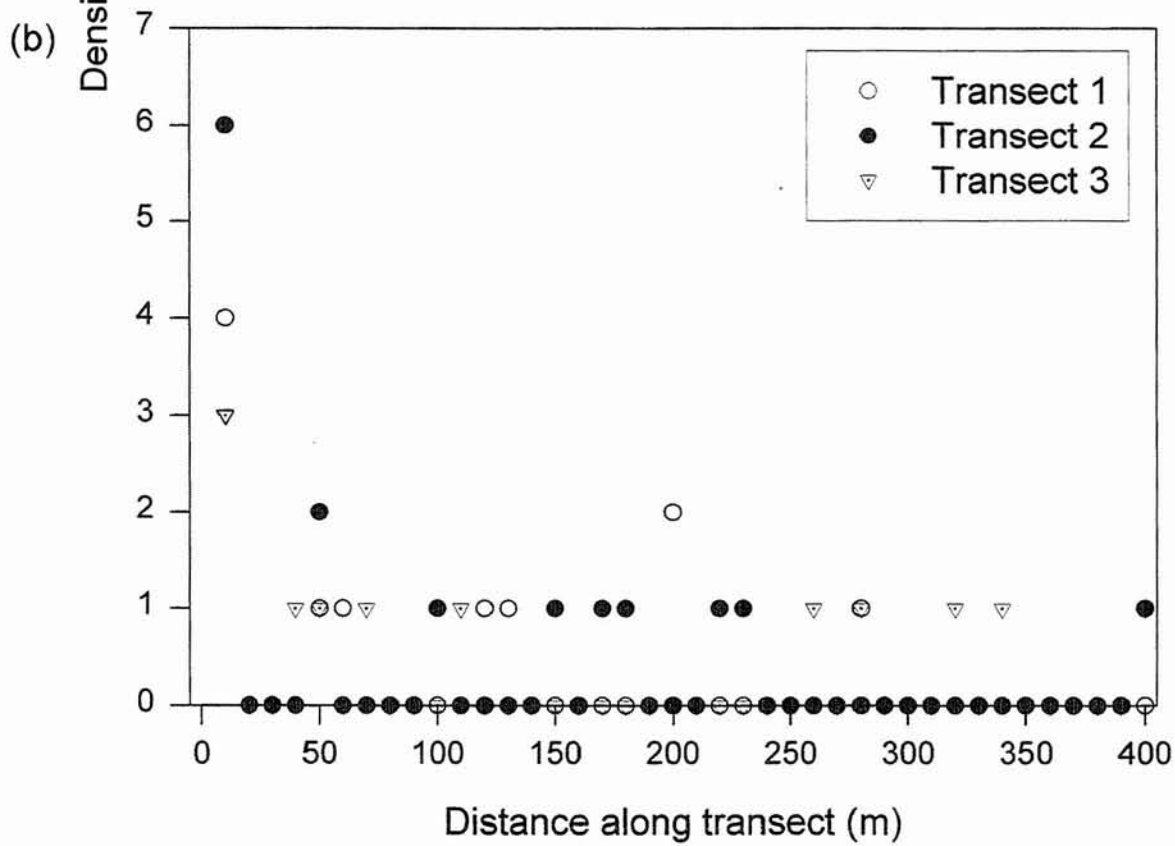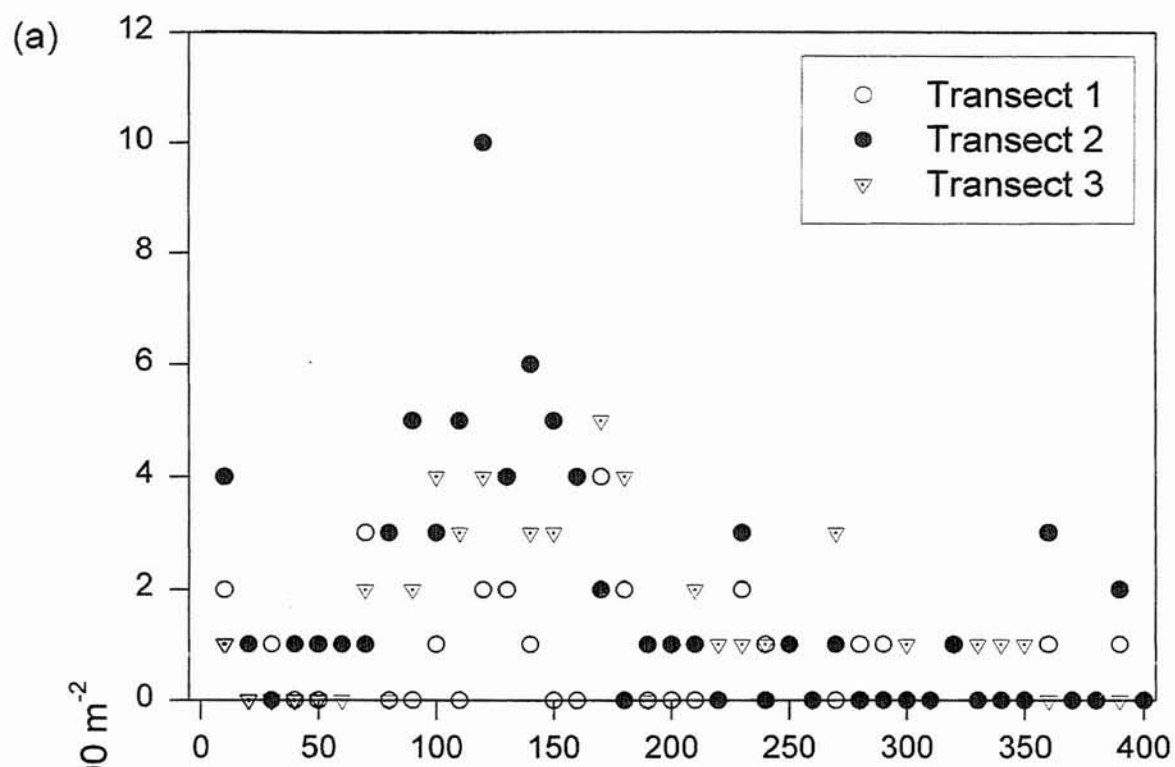**Figure 6.4**   Distribution of *A. mangium* densities (100 m^{-2}) on each of the three transects at Site 3.

(a)   Individuals > 1.5 m

(b)   Individuals < 1.5 m
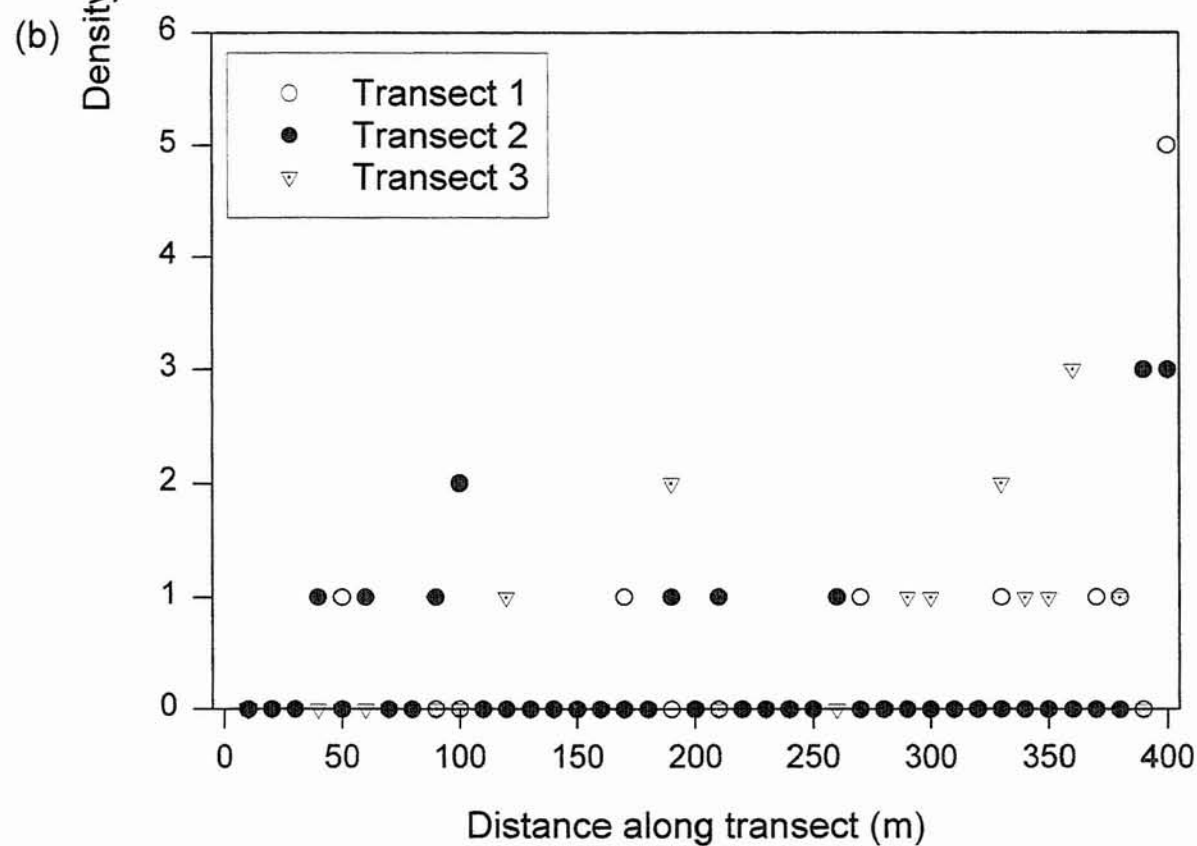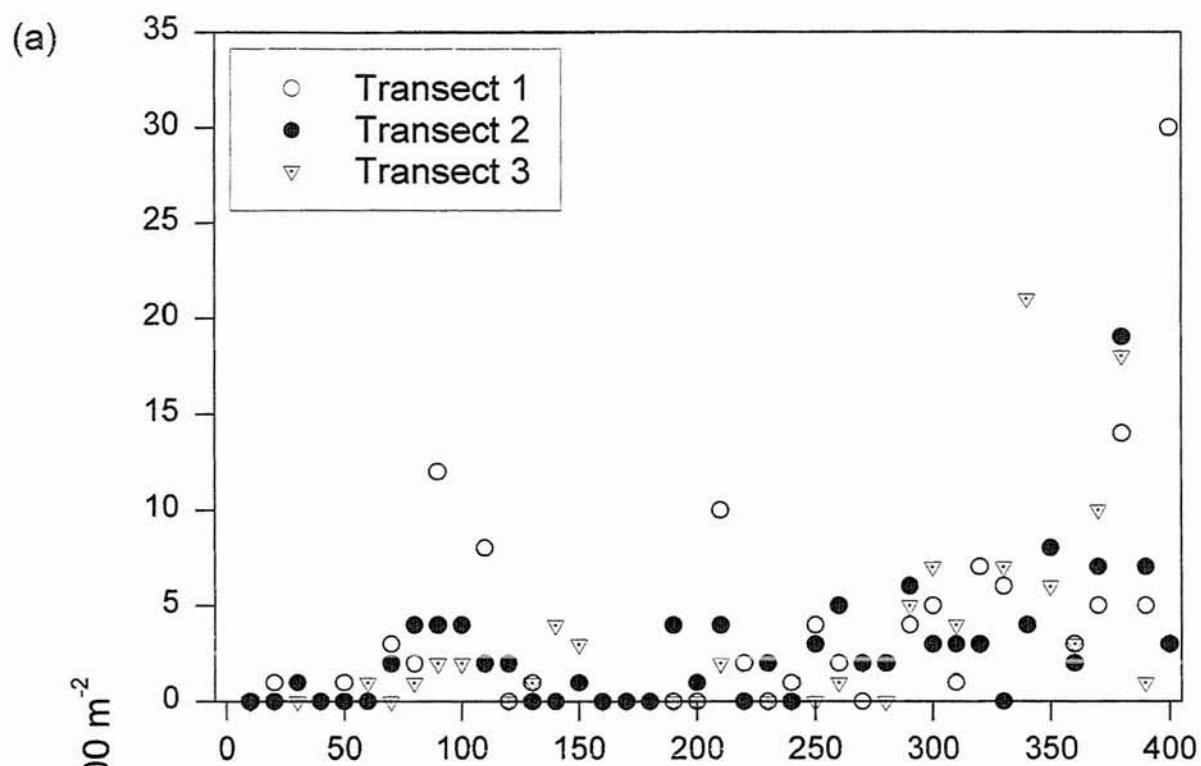
(a)

(b)

Density 100 m^{-2}

Distance along transect (m)

Figure 6.5   Growth measurements of *P. caribaea* var. *hondurensis* taken along Transect 2 at Site 3.

(a)   Mean height (m) and DBH (cm) (100 m^{-2}) (± SE)
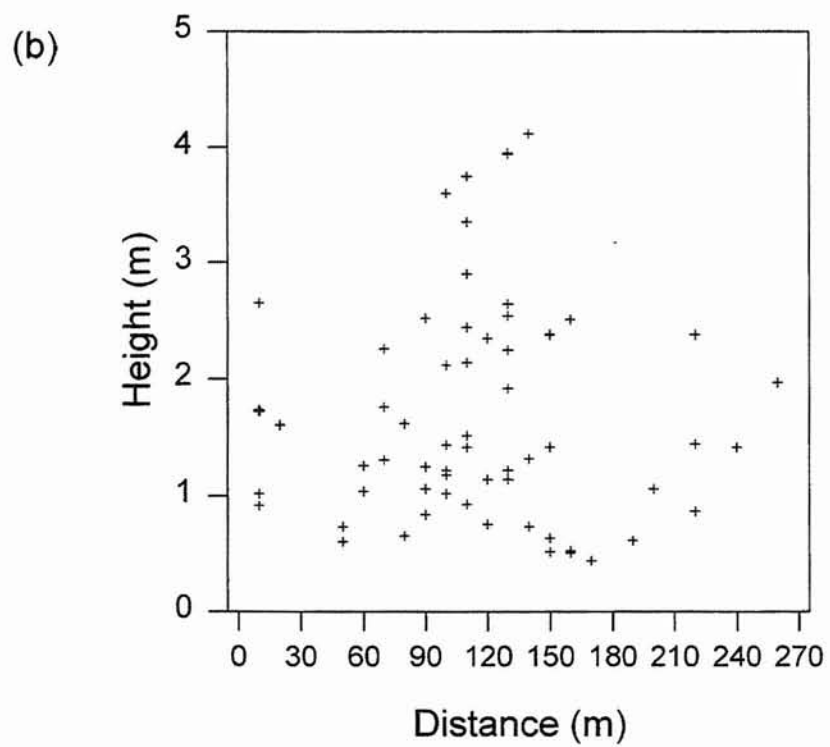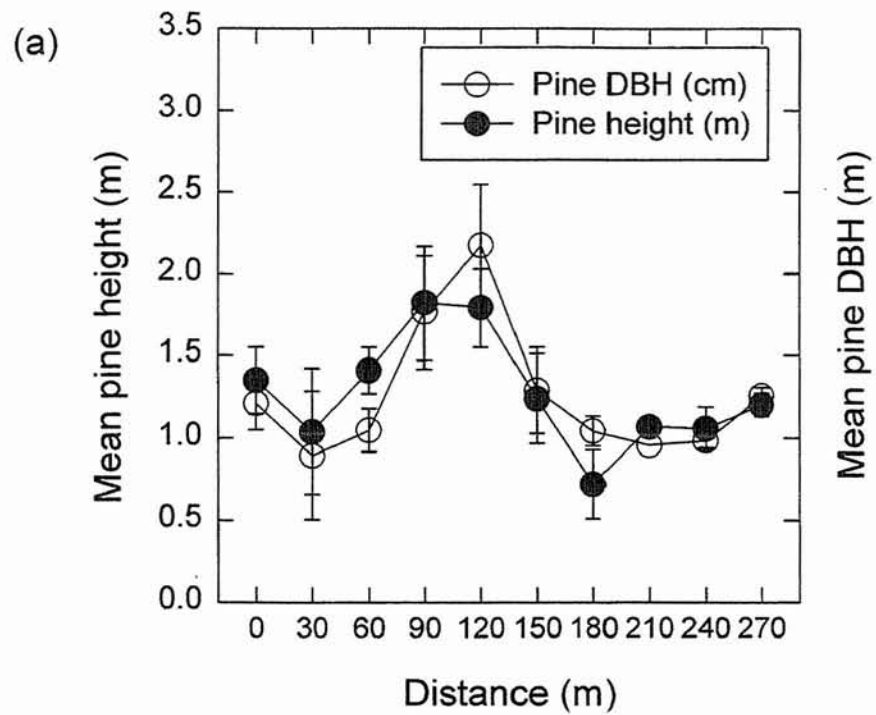
(b)   Height (m) distribution data

(a)

(b)

Table 6.2   Multiple linear regression and ANOVA results for pooled pine distribution data (i.e. total pine density = a + b*(total density of other species), where b = slope and a = intercept of the regression line). Significant results are highlighted in bold.

Variable	Regression (r^2=0.28)			ANOVA	
(Species totals)	b	T	p	F ratio	p
Acacia mangium	-0.92	-2.94	**0.004**	14.75	**0.000**
Acacia crassicarpa	0.12	2.77	**0.007**		
Casuarina equisetifolia var. *incana*	0.16	4.21	**0.000**		
Constant (a)	1.5	7.42	**0.000**		

Table 6.3   Multiple linear regression and ANOVA results for the distribution of pines >1.5 m (i.e. pine density >1.5 m = a + b*(total density of other species), where b = slope and a = intercept of the regression line). Significant results are highlighted in bold.

| Variable	Regression (r^2=0.16)			ANOVA	
	b	T	p	F ratio	p
Acacia mangium (<1.5 m)	-0.08	-2.69	**0.008**	11.35	**0.000**
Acacia crassicarpa (<1.5 m)	0.34	4.06	**0.000**		
Constant (a)	1.16	6.19	**0.000**		

Table 6.4   **Multiple linear regression and ANOVA results for the distribution of pines <1.5 m (i.e. pine density < 1.5 m = a + b*(total density of other species), where b = slope and a = intercept of the regression line). Significant results are highlighted in bold.**

Variable	Regression (r^2=0.58)			ANOVA	
	b	T	p	F ratio	p
Casuarina equisetifolia var. *incana* (<1.5 m)	0.14	4.05	**0.0001**	81.65	**0.000**
Casuarina equisetifolia var. *incana* (>1.5 m)	0.15	9.35	**0.000**		
Constant (a)	0.21	4.33	**0.000**		

The growth rate (height and DBH increases) results for both species are presented in Figure 6.6. *Pinus caribaea* var. *hondurensis* always recorded larger increases in both height and DBH regardless of the time of year, with figures sometimes 20 times higher than those of the *Melaleuca*. Some *M. viridiflora* individuals showed no growth increase over the 22 months of the study indicating that the plants were in poor condition. By June 1995, 37% of the *M. viridiflora* plants had died, whilst no mortality was recorded for any of the juvenile pines. Both repeated measures analyses indicated that the differences between the two species were always significant, and these findings were supported in both cases by the subsequent uni-variate analyses provided by this technique (Tables 6.5 and 6.6). An analysis of sphericity indicated that the assumptions of the ANOVAs had not been violated.

The prescribed fire at the transect site was relatively intense and resulted in a 58% reduction in the numbers of *M. viridiflora* individuals in the 1.5 to 3.0 m height size class, a 97% reduction in fine fuel and scorch heights of 4 to 5 m (Chapter 5). These values are higher than those recorded for any other prescribed burn in *M. viridiflora* woodlands examined during this study. Scorch heights were highest at the northern end of the transects, indicating that the fire became hottest as it reached the closed forest margin.

Figure 6.6   Mean height (cm) and DBH (mm) (± SE) changes for *M. viridiflora* and *P. caribaea* var. *hondurensis* at Site 3.

Mean change (Δ) in height (cm)

Mean change (Δ) in diameter (m)

Time period (months)
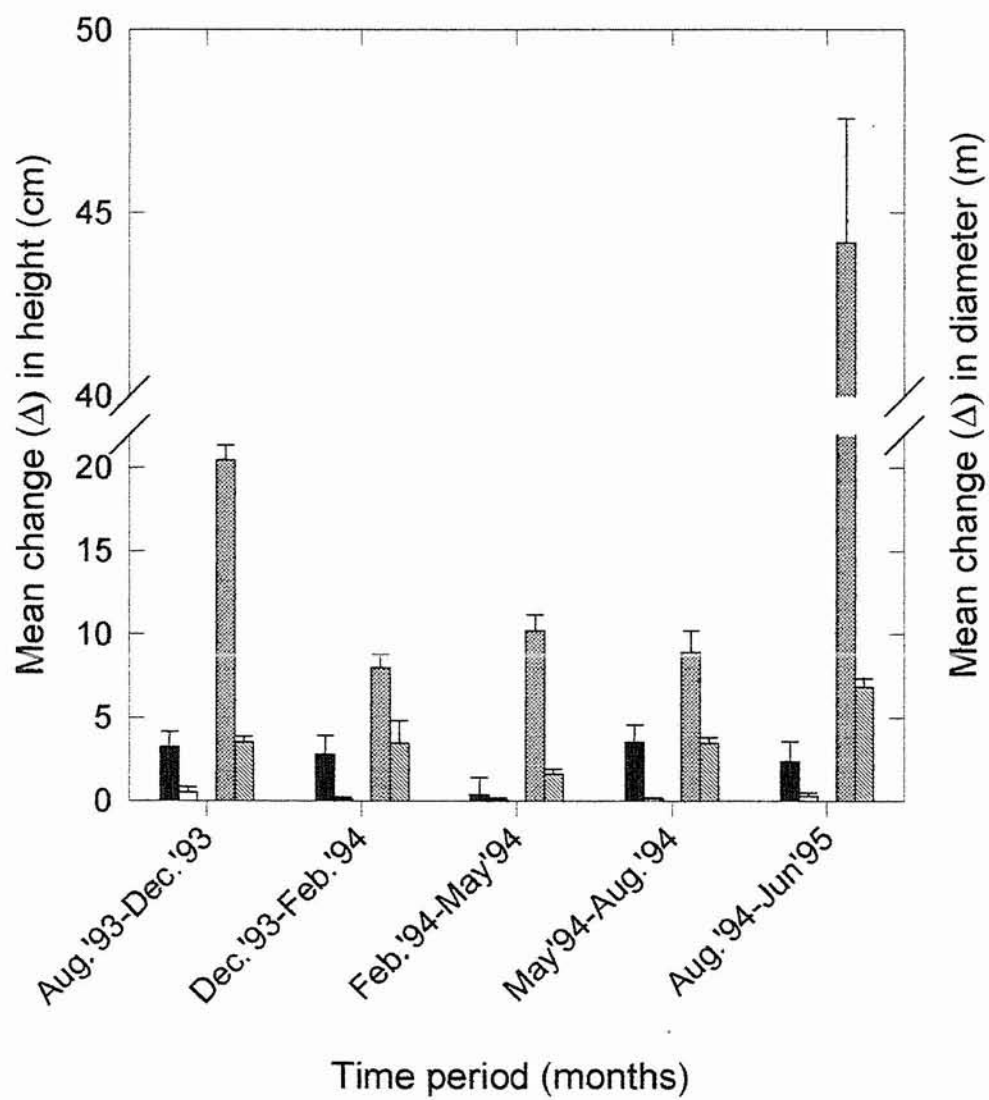
M. viridiflora-Δ height

M.viridiflora-Δ diameter

P. caribaea var. hondurensis-Δ height

P.caribaea var. hondurensis-Δ diameter

Table 6.5   Results from the repeated measures ANOVA of the height data for both species over the duration of the study, showing F ratios and their significance. Significant results are highlighted in bold.

Variable/Interaction	F ratio	df	p value
Multi-variate			
Time	50.41	4	**<0.0001**
Time*species	46.33	4	**<0.0001**
Between subjects	167.25	1	**<0.0001**
Uni-variate			
Within subjects (time)	70.18	4	**<0.0001**
Within subjects (time*species)	67.27	4	**<0.0001**

Table 6.6   Results from the repeated measures ANOVA of the DBH data for both species over the duration of the study, showing F ratios and their significance. Significant results are highlighted in bold.

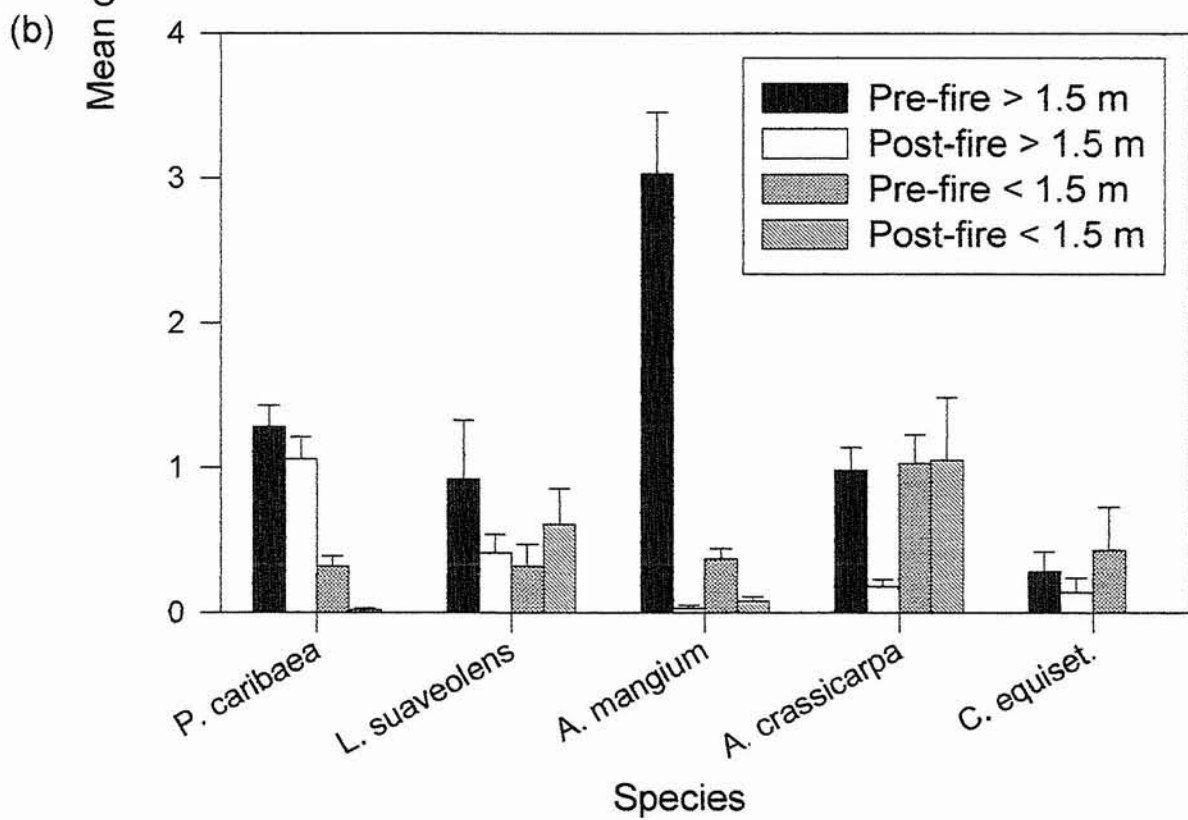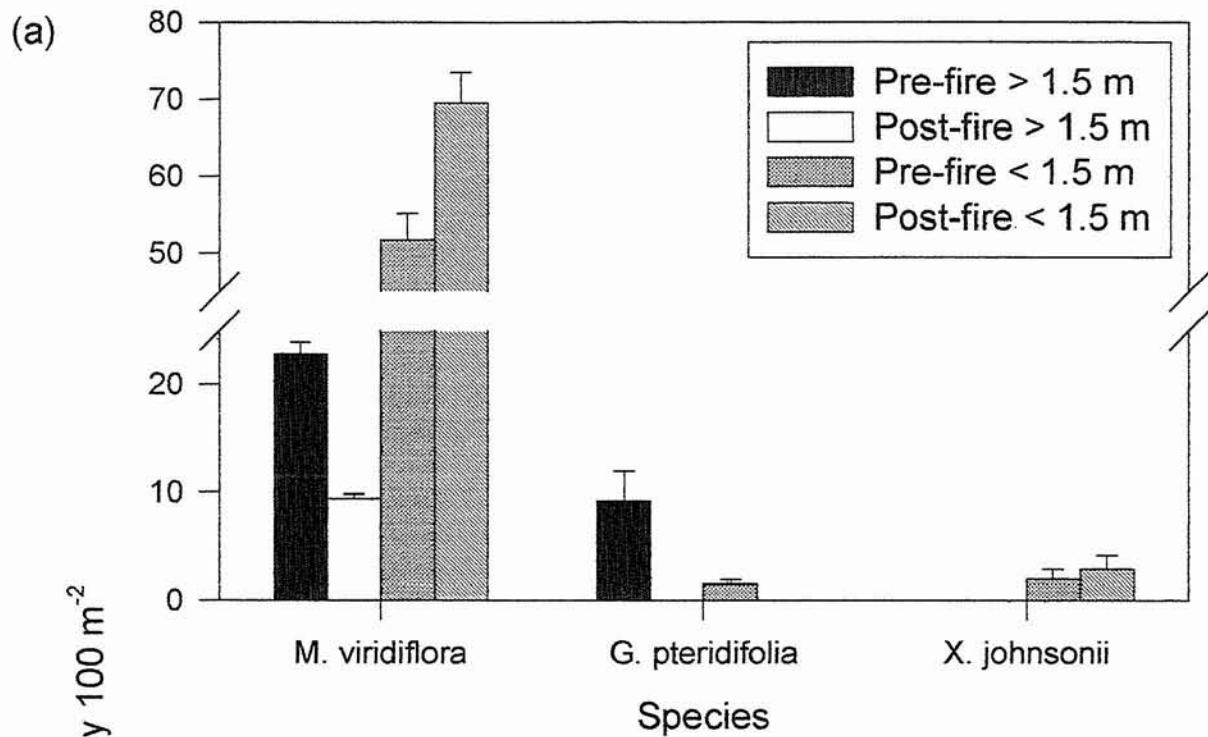| Variable/Interaction	F ratio	df	p value
Multi-variate			
Time	21.16	4	**<0.0001**
Time*species	17.77	4	**<0.0001**
Between subjects	292.76	1	**<0.0001**
Uni-variate			
Within subjects (time)	25.64	4	**<0.0001**
Within subjects (time*species)	19.53	4	**<0.0001**

For the eight species examined, the number of individuals in each size class (pre- and post-fire) is summarised in Figure 6.7. The statistical results from the one-way ANOVA for each species (size classes pooled) are presented in Table 6.7. Three species (*P. caribaea* var. *hondurensis*, *Acacia mangium* and *Grevillea pteridifolia*) recorded a significant difference in total numbers following fire (Table 6.7). Fire almost completely killed *P. caribaea* var. *hondurensis* individuals less than 1.5 m tall, whilst little effect was apparent on larger trees (Figure 6.7b). Similar, although not significant, trends were apparent for *Casuarina equisetifolia* var. *incana* (Figure 6.7b). The two other species that had significant changes in numbers recorded either substantial reductions (*A. mangium*) or 100% loss in both size classes (*G. pteridifolia*) (Figure 6.7a-b).

Table 6.7   Uni-variate ANOVA results for the effects of the prescribed fire on total species density, showing F ratios and their significance.   Significant results are highlighted in bold.

Species	F ratio	p
Melaleuca viridiflora	0.51	0.48
Pinus caribaea var. *hondurensis*	4.26	**0.04**
Lophostemon suaveolens	0.11	0.75
Acacia mangium	51.86	**0.00**
Acacia crassicarpa	2.01	0.16
Casuarina equisetifolia var. *incana*	2.09	0.15
Grevillea pteridifolia	11.73	**0.001**
Xanthorrhoea johnsonii	0.31	0.58

Figure 6.7   The effect of fire on the density (100 m^{-2}) ($\pm$ SE) of eight species in two size classes (> 1.5 m and <1 .5 m) at Site 3.

    (a)   Species with relatively high mean densities

    (b)   Species with all average densities less than five

(a)

Mean density 100 m^{-2}

Pre-fire > 1.5 m
Post-fire > 1.5 m
Pre-fire < 1.5 m
Post-fire < 1.5 m

M. viridiflora   G. pteridifolia   X. johnsonii

Species

(b)

Pre-fire > 1.5 m
Post-fire > 1.5 m
Pre-fire < 1.5 m
Post-fire < 1.5 m

P. caribaea   L. suaveolens   A. mangium   A. crassicarpa   C. equiset.

Species

As with the other fires documented in this study (Chapter 5), *M. viridiflora* exhibited an increase in the number of sprouts and saplings present in the understorey (individuals <1.5 m) and a loss of larger saplings. Similarly, as previously documented, *X. johnsonii* increased in abundance following fire. In terms of total abundance, however, no significant effect was detected for either species (Table 6.7). Comparable trends were recorded for *Lophostemon suaveolens*. *Acacia crassicarpa* was the only species that exhibited a marked reduction in individuals taller than 1.5 m, whilst juveniles remained essentially unchanged (Figure 6.7b).

6.3.2   Germination experiments

Results of the temperature and moisture germination experiments are presented in Figures 6.8a-b respectively. *Melaleuca viridiflora* seeds exhibited higher numbers of seedlings at all temperatures except 200 and 600°C (Figure 6.8a). At these high temperatures neither species recorded any germination over the 28 day period. In addition, *M. viridiflora* continued to germinate (although in reduced numbers) at temperatures where no *P. caribaea* var. *hondurensis* germination was recorded (100 and 150°C). The ANOVA indicated that the germination responses of the two species were significantly different with respect to temperature (Table 6.8).

Melaleuca viridiflora seeds also yielded higher numbers of seedlings (after 28 days) than the pine for all moisture treatments except zero where no germination of either species was documented (Figure 6.8b). The germination of *P. caribaea* var. *hondurensis* decreased sharply at moisture levels greater than 25%, whilst *M. viridiflora* showed similar levels of germination for moisture treatments ranging from 10% to 40%. As with the temperature experiment, the ANOVA indicated that the germination responses of the two species were significantly different with respect to moisture, although the differences were less pronounced (Table 6.8).

Figure 6.8   Mean number of seedlings (± SE) of *M. viridiflora* and *P. caribaea* var. *hondurensis* following 28 days of different temperature and moisture treatments.

      (a)   **Temperature effects**
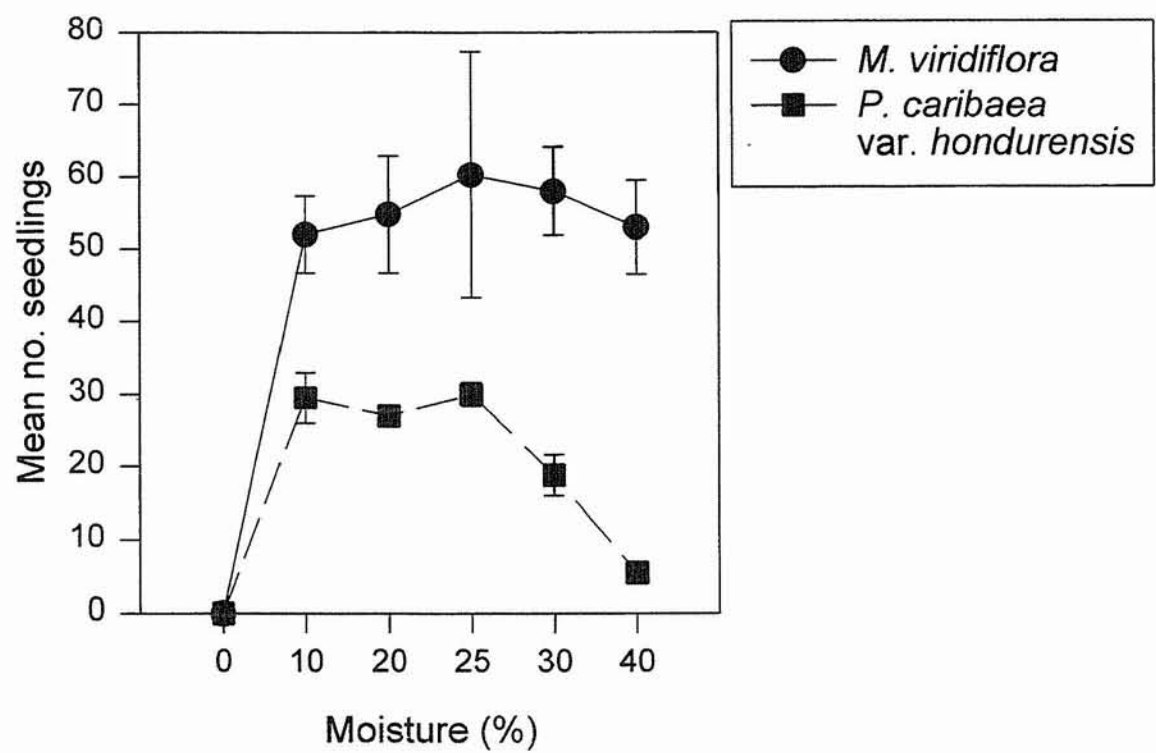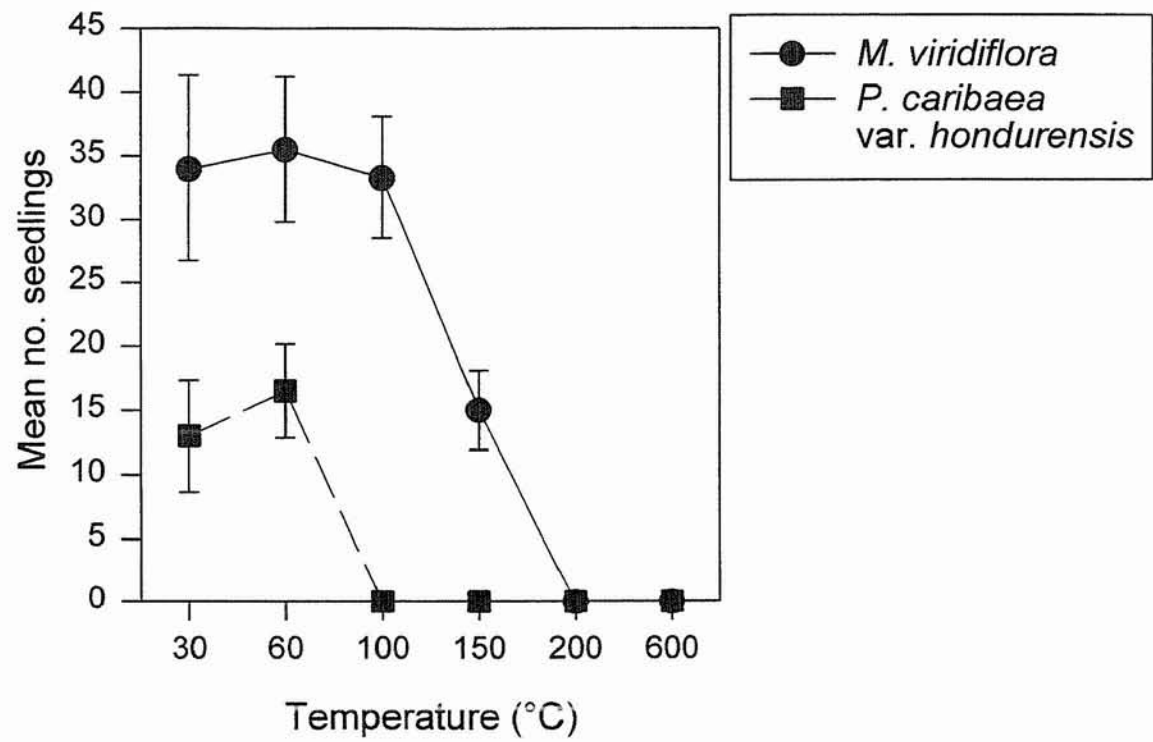
      (b)   **Moisture effects**

Table 6.8   **Results of two-way ANOVAs of temperature and moisture, showing F ratios and their significance.** Significant results are highlighted in bold.

Interaction	d.f.	F ratio	p
Species*temperature	5	6.67	**<0.001**
Species*moisture	5	3.21	**0.02**

6.4   DISCUSSION

Pinus caribaea var. *hondurensis* has successfully invaded a remnant *M. viridiflora*-dominated open woodland community less than 20 years after planting. The distribution of the predominant native species only explained, at best, 58% of the variation observed in pine tree distribution (Table 6.4). Interestingly, the native tree with a growth form closest to that of the exotic (*C. equisetifolia* var. *incana*) best explained the observed distribution of small (<1.5 m) pine trees (Table 6.4). The overall lack of relationship between the distribution of native species and pines suggests that the exotic is a very aggressive coloniser of the woodland community, and is not restricted to certain areas where favourable species suites (in terms of critical compositions and/or densities) occur.

Densities of the exotic tree are, however, highest at two locations on the transects studied (Figure 6.3). Firstly, pines are abundant at the edge of the community closest to the plantation. This is not unexpected because the edge of the woodland forms the first effective barrier to seed dispersal, and weed invasion has repeatedly been demonstrated to be most prolific at the edges of remnant plant communities in Australia (e.g. Saunders *et al.* 1991). Secondly, pine density peaked between 80 to 180 m, where there was no observable changes in topography, soil type or grass cover. The wide range of pine heights and DBHs in this region (Figure 6.5) imply that more than one successful seeding event has occurred. This area may reflect the most likely dispersal distances under the prevailing wind conditions at the time of seed release. Variations in wind strength, rain intensity and seed weight may account for extensions and reductions in these distances. Additionally, some seed is likely to be bird-dispersed (Richardson & Bond 1991), which

could also explain some of the variation recorded in the dispersal distances.

The peaks in pine density may reflect the occurrence of more favourable site conditions such as increased light, moisture, and nutrient availability. Reductions in competition and predation pressure may also be important. Competition in the regeneration niche (Richardson & Bond 1991) is likely to be low given the absence of any other fast-growing, shade-producing and self-pollinating species. Testing any or all of these hypotheses could form the basis of future research at this site.

It would also be of interest to determine whether competition (similar to that reported in temperate regions with eucalypts) is developing between the pine and *M. viridiflora*. This is highly likely given that the pine has growth (and survival) rates many times higher than those of the native tree (Figure 6.6). Additionally, the development of large pine needle blankets in the understorey and high levels of shade in an otherwise open woodland habitat are not likely to favour the establishment of *M. viridiflora* (or other native species). Competition is also likely as the two species release seed at similar times, even though seed longevity is predicted to be low for this species of pine under field conditions (Dr M. Dieters, pers. comm.).

Fire had a significant effect on the density of pines in the woodland habitat (Figure 6.7 and Table 6.7). Juvenile pine trees were particularly susceptible as was *A cacia mangium*. This large scale (12,000 m^2) study supported the effects of fire documented for *M. viridiflora* woodlands at other sites which had smaller (500 m^2) sample areas (Chapter 5). The density of *M. viridiflora* juveniles (pre-fire) was highest at the northern end of the transects where the fire in this study became hottest. Perhaps past fires have also been most intense here, resulting in prolific germination and sprouting of *M. viridiflora*. This type of response has been documented following the other fires investigated during this thesis (Chapter 5).

The high temperatures associated with even moderate intensity fires in these woodlands are likely to have a significant effect on pine seed germination. *Pinus caribaea* var. *hondurensis* did not germinate above temperatures of 60°C (Figure 6.8), and the

susceptibility of both seed and juveniles of this species to fire has already been reported (Dr M. Dieters, pers. comm.). It is therefore possible that back fires (usually reaching high fire temperatures) may differentially affect seed survival of the two species, and hence be a useful option available to management authorities. Unfortunately, however, intense fire at the transect site did little to reduce the density of pine trees over 1.5 m in height. Many of these individuals resprouted prolifically following fire even if they had been severely scorched. Fire is therefore not seen as a panacea for pine control in *M. viridiflora* woodlands, probably only being effective for reducing small trees and seed germination if managed correctly.

Fire may in fact create as many problems as it solves. Firstly, many *M. viridiflora* communities located on the mainland (particularly those close to roads) are already burnt too frequently. Research from this thesis (Chapter 5) has outlined the need for fire frequencies in the order of one fire every ten years. This frequency is not likely to control even juvenile pines as they can grow to a size at which they are resistant to fire within 12 months of establishment. Secondly, fire may also create ideal conditions for pine seed germination (Cronk & Fuller 1995). Fires in the wet tropics region have already been shown to facilitate weed invasion, which remains a concern of land management agencies (Mr P. Stanton, pers. comm.). Finally, low intensity fires which can occur earlier in the year (and do not burn the entire understorey) are less likely to affect juvenile establishment. Furthermore, they may precede seed release and thus be of less use for the control of this exotic species.

Pine trees were recorded 400 m into the woodland community, a total dispersal distance (including the fire break) of nearly 500 m. Many significant remnant *M. viridiflora* woodlands are now located either adjacent to or not far from pine plantations (see Figure 2.2). Many are also similar in size or smaller than the communities studied here. It is therefore anticipated that pine invasion into other *M. viridiflora* communities close to plantations is highly likely at some time in the future, once the pines have reached reproductive maturity (many of the plantations in the region to the south of the study site are less than five years old). Additionally, current invasions are only likely to worsen as pines:

- modify the existing woodland habitat to essentially create conditions more favourable for themselves;

- continue to invade from an abundant seed source; and

- already established in *M. viridiflora* woodlands reach maturity, begin to self-pollinate, release seed and thus contribute their own progeny to the woodland community.

Thus there is great potential for the invasion process to become more widespread with subsequent reductions likely in *M. viridiflora* habitat quality and biodiversity. The susceptibility of this community to invasion following disturbance has long been recognised (Tracey 1982). To date, however, like many other tropical lowland plant communities, *M. viridiflora* woodlands have received little management or conservation attention.

Melaleuca viridiflora communities threatened by pine invasion include those within reserves currently afforded the highest possible conservation status in Australia (i.e. the WTWHA). The formulation and implementation of management plans for this species are urgently required so that the invasion issue is acknowledged by all management agencies. Management of invasions only becomes more costly as the level of infestation increases, and late intervention usually ensures that a reduction in the conservation values of the remnant community has already occurred (Hobbs & Humphries 1995). An integrated approach to control using a combination of methods including hand removal, poisoning and fire (later in the year after seed release) may provide the best management solution. Finally, to reduce the future potential for invasions, forestry research programs could examine the possibility of developing sterile plantation species.

CHAPTER SEVEN - CONCLUSIONS, IMPLICATIONS AND

FUTURE RESEARCH OPPORTUNITIES

The past is for wisdom and the future for action (Anonymous).

The pressure upon remnant vegetation in Australia is arguably at its most intense. This stems from, amongst other factors, on-going clearing of native vegetation for agricultural and urban expansion, exotic species invasions and inappropriate and/or unmonitored fire management practices. These broad-scale impacts are considered highly significant in the coastal regions of the country where clearing has been most extensive, and habitat diversity and the proportion of threatened species are highest (Thomas & McDonald 1989; Ingram & Raven 1991; DEST 1995a). The lowland coastal sections of the WTBR are no exception, with many habitats (and species within them) considered endangered or vulnerable as a result of inadequate representation in the existing conservation reserve system (QDEH 1995a).

These assessments of conservation status are, however, currently made with little quantitative knowledge of past clearing patterns. Additionally, no quantitative data exists regarding the type or extent of habitat change associated with past alterations to disturbance regimes. Thus, the actual conservation status assigned to vegetation types may, at best, be approximate and, at worst, inaccurate. This thesis therefore initially focussed on providing an accurate account of past habitat clearing, fragmentation and habitat change in a lowland section of the WTBR widely recognised for its ecological importance (Webb 1966; Stanton & Morgan 1977; Stanton & Godwin 1989; Lavarack 1994; Tait 1994; Blackman *et al.* 1996). Clearing rates much higher than those previously calculated were reported. Additionally, it was confirmed that successional changes in some remnants were occurring (conversion of open woodlands to forests because of a lack of fire) to the detriment of habitat diversity. This type of habitat change as a result of altered fire regimes had only ever been recorded for wet sclerophyll forests adjacent to rainforests (Harrington & Sanderson 1994), and in grasslands on Cape York Peninsula (Crowley & Garnett 1995).

Future research following on from the analysis of clearing/fragmentation could target specific fragments selected on the basis of vegetation type, remnant size and shape, position in the landscape, land tenure, fire history and any other variables of interest to land managers.

These remnants could then be assessed in terms of species richness, structural integrity, suitability for inclusion in the conservation reserve system and susceptibility to conversion or exotic species invasion. Where remnants are degraded, the effects and effectiveness of various rehabilitation techniques and management initiatives could be tested. Additionally, the impact and management of other significant disturbances affecting remnants which were not examined during this study (including altered hydrological patterns resulting from stream diversion and drainage) could be investigated.

The information already obtained from the assessment of clearing and fragmentation has enabled management agencies to:

· identify critical remnant vegetation types and habitats for inclusion in either the existing conservation reserve system or "off-park" conservation reserves; and

· assign priority to management initiatives (the use of prescribed fire) designed to reverse losses of habitat diversity.

The ecologically sustainable management of remnant habitats (irrespective of land tenure, position in the landscape, and in some cases even size) is widely touted as being of the utmost importance for the conservation of biodiversity. Whilst this largely intangible goal of government policy is undoubtedly desirable from a conservation perspective, management agencies usually lack even baseline ecological information (including community composition and structure, and responses to disturbances) to facilitate the formulation of appropriate management plans and practices. This is certainly true for the vegetation types of the highly diverse tropical lowland habitat mosaic, where a relative scarcity of ecological research has been conducted. As a result, management agencies are forced to implement "best-guess" management practices with little or no understanding of the effects of such practices on ecosystem processes and biodiversity. The secondary focus of this thesis was, therefore, a detailed assessment of the structure and composition of a threatened lowland plant community (*M. viridiflora* open woodlands) throughout its distribution in the WTBR. In addition, information relevant to two critical management issues currently placing pressure on these woodlands (fire management and exotic species invasion) was obtained. The implications of these research findings are discussed below in the context of sustainable

management of *M. viridiflora* woodlands and potential areas for future research.

Fire management of remnant vegetation remains a highly controversial topic in Australia for a wide variety of reasons (Williams & Gill 1995). Central to the controversy is the lack of available information on conceivably the most important ecological disturbance affecting terrestrial ecosystems. This problem has been lamented by land managers, conservationists and researchers alike (Stanton 1992; Nias 1995; Gill *et al.* 1996), and remains a serious issue despite repeated efforts to develop a minimum data set for monitoring the effects of fire on conservation reserves (Gill & Nichols 1989; Wardell-Johnson *et al.* 1989; Warriner-Lawrence 1993). Unfortunately, no national standards have been adopted, a concern raised with respect to other fire management research issues in Australia (McCaw 1991; Chapter 5 this study). The need for national standards across a wide range of fire management research disciplines is urgent, as is adequate funding to implement sufficiently detailed monitoring programs in all conservation reserves.

The lack of information regarding the effects of fire in the WTBR has partially been rectified following attempts to implement a standardised monitoring system across the north-eastern national park estate (Skull 1994b; Bell 1995; Williams 1996). The final program that has been adopted is, however, relatively simple, and lacks adequate levels of replication for detailed scientific analysis. This reflects not only a lack of resources, but also the skills of parks staff who are responsible for collecting the data. Thus there is also a need for long-term, large-scale fire research programs to be initiated so that the effects of fire regimes (rather than single fires) can be analysed. Other researchers have also recognised the need for these types of studies (Whelan 1995; Williams & Gill 1995). Only a few, however, have been conducted in Australia (e.g. research programs at Munmarlary and Kapalga in the Northern Territory), none of which has focussed on the WTBR. This is despite the fact that fire management plans are now in place for conservation reserves of national significance throughout the region. The need for localised studies is paramount, given that the transfer of fire research results and management practices between regions is generally not recommended, even at the individual species level (Williams *et al.* 1994, Chapter 5 this study).

Management agencies within the WTBR are, however, compelled to act now, unable to wait for research to provide management recommendations for all of the vegetation types (let alone species) in the region. Many remnant habitats now occur as isolated islands surrounded by a highly disturbed landscape, and massive habitat changes are taking place (Stanton 1992, Chapter 2 this study). Failure to implement fire management strategies for remnant communities is likely to result in further declines in habitat diversity and, ultimately, biodiversity. The appropriate timing, frequency and intensity of prescribed burning programs to ensure sustainable management remain essentially unknown for many vegetation types.

The use of prescribed fire to maintain habitat diversity has been demonstrated to be effective in terms of re-establishing open woodland communities invaded by more mesic species associated with closed forests (QNPWS 1991). However, its use in this manner remains highly controversial in north-eastern Queensland, although little information on the debate has been published. The argument against its use stems from the high values the community associates with closed forests (especially rainforests), and a desire to see landscapes return to their pre-European or even pre-Aboriginal state. Unfortunately, the proponents of this argument have a poor understanding of:

- the lack of concrete knowledge regarding the condition of the landscape prior to the settlement of either Europeans or Aboriginals;
- the massive changes in vegetation patterns across the landscape (from clearing and fragmentation), fire regimes and exotic species that make any return to past conditions or practices unlikely and even undesirable;
- the ecological significance of the habitats being replaced;
- the rapid pace of the change involved; and
- the largely irreversible nature of the process once it has occurred.

It must be decided whether the maintenance of habitat diversity is an acceptable goal of natural resource management. It may be that when (and if) research data becomes available, this approach may not be effective, entirely achievable or appropriate in some situations. It does, however, "buy time", providing the opportunity to undertake research so that managers of the future can make decisions based on scientific data. At the same time it allows for the

maintenance of significant components of lowland habitat diversity. The argument against this type of management is fundamentally flawed for two major reasons. Firstly, some of the communities threatened by successional change/conversion are so endangered, that the few remnants located within highly modified landscapes must be protected. Secondly, the speed of the conversion (even if the end result of the change is accepted) is too high for some species (which may have survived a more prolonged change) to adapt to the changing environment.

This discussion to date has only related to habitats undergoing the conversion process outlined above. The remaining discussion on fire management in this section relates specifically to the *M. viridiflora* community studied during this investigation which, as previously discussed (see Chapter 5), is not subject to this process. This study and others (Jackson 1968; Braithwaite & Estbergs 1985; Bradstock 1990; Tolhurst 1996; Bond & van Wilgen 1996) have shown that repeated fires (with short fire-free intervals) can have profound effects on community structure and recruitment. The resultant simplification of community structure represents a potential threat to the diversity of both flora and fauna.

The simplification of stand structure was documented at Site 17, known to have been subjected to an annual fire regime in recent times. Following a change in land ownership, this regime has been interrupted by management agencies, with a view to leaving the site unburnt for several years. This site therefore provides a unique opportunity to document the response of the community to a relatively long fire-free period. Of particular research interest would be:

- the effects the altered fire regime has on community structure and composition (how do these parameters compare with a site where the regime is continued?);
- the role of the seed bank (does any viable seed remain?) in regeneration at the site;
- the ability of individuals which have been reduced to resprouting lignotubers by repetitive fires to re-establish and eventually reach reproductive maturity (have lignotuber reserves been exhausted?); and
- the success of any new cohorts of seedlings (has the frequent fire regime affected the regeneration niche?).

The QDE have tentatively adopted the findings of this research and adjusted fire management plans (e.g. QDEH 1996a) in several ways. Firstly, fire frequencies prescribed for *M. viridiflora* woodlands within mainland conservation reserves have been altered to more conservative levels (from one in two to three years as a minimum to one in five to ten years). This range of frequencies allows for comparisons to be made between fire regimes recommended for the maintenance of structural integrity and those proposed for orchid population conservation (Lavarack 1994). Secondly, some areas have been set aside to exclude fire altogether, which will allow comparisons to be made with sites on HINP that have been burnt on a ten yearly basis. Finally, wherever possible, annual fire regimes will be avoided.

Unfortunately, the other significant management issue relevant to *M. viridiflora* woodlands documented during this study (the invasion of exotic pine into vulnerable remnant habitats) has not been acknowledged by the appropriate government agency. This is despite the accepted view that exotic species invasion is a major threat to biodiversity in Australia, with many localised plant extinctions reported as a result (Adair 1995; Hobbs & Humphries 1995). The pine invasion problem is also likely to worsen at sites where it has been initiated once individuals established within the woodlands mature and begin to contribute their own progeny. The problem is also likely to spread to currently unaffected sites as recently planted pine communities reach reproductive maturity. This previously undocumented invasion of Caribbean pine presents further opportunities for research including:

- an assessment of whether competition develops between the dominant native species and the exotic tree. This type of competition has been reported by the only published long-term study of pine invasion in Australia (Burdon & Chilvers 1994);
- examining alterations to fuel loads and flammability of *M. viridiflora* communities;
- investigating possible changes in species richness and composition; and
- comparing patterns of invasion at other habitats when the process is first initiated rather than after the initial invasion event.

There is, however, an urgent need to focus on control issues. The use of prescribed fire has been shown by this and studies on other pine species (e.g. Richardson & Bond 1991) to be

an ineffective technique for the control of larger trees. Manual removal, although considered to be the most ecologically acceptable management option (Bradley 1988), is only useful for the control of juveniles. Future research could also target developing the most effective and cost-efficient combination of control techniques for invaded sites. If routine inspections and, where necessary, removal of juveniles, were to be incorporated into current plantation management policy, then the invasion problem could be minimised. The importance of early intervention (in terms of the costs associated with control) has already been highlighted by other researchers (Hobbs & Humphries 1995).

This study has provided valuable quantitative information on the past pattern of clearing and fragmentation in the WTBR. It is possible that clearing rates in other areas of the region are higher, given that they have been subject to very extensive sugar cane development (Ms J. Kemp, pers. comm.). Furthermore, clearing in some sections of the area examined has not halted, so the figures reported from the 1992 assessment are now likely to underestimate habitat loss. This research has also provided the first detailed assessment of the structure of *M. viridiflora* woodlands in the WTBR, documented their responses to fire and highlighted other important management issues. There is great scope for further research, including increasing the number of sites examined to gain an improved understanding of the relationship between soil type, fire history and structural diversity. This should be coupled with further investigations into the effects of fire on community structure and composition. There is also an urgent need for all management agencies to collectively address management issues relevant to this and other plant communities associated with the tropical lowland habitat mosaic. To facilitate the conservation of biodiversity, this needs to be conducted in an integrated fashion and at the landscape level. Only then will it be possible to ensure the ecologically sustainable development and management of these highly significant remnant communities.