final report

Project Code:	NAP3.206
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Date published:	November 1999

PUBLISHED BY
Meat and Livestock Australia Limited
Locked Bag 991
NORTH SYDNEY NSW 2059

Managing Tropical Woodlands to Control Exotic Woody Weeds

Meat & Livestock Australia acknowledges the matching funds provided by the Australian Government to support the research and development detailed in this publication.

TABLE OF CONTENTS

MANAGING TROPICAL WOODLANDS TO CONTROL EXOTIC WOODY WEEDS

INTRODUCTION

Background

A suite of exotic woody weeds threatens the ecological and economic sustainability of Australian tropical rangelands. The ability of the pastoral industry to tackle this problem is limited by several factors. First, it affects a very large area already and there is enormous potential for most of these weeds to spread much further. Second, the threat is from a range of species that differ considerably in their basic biological characteristics and in their responses to control measures, and yet, several species may co-occur. Third, there are severe economic constraints within the pastoral industries that are threatened by exotic woody weeds. Together, these factors demand that the extremely limited resources available for addressing woody weed problems be used as efficiently as possible. The timing of actions to counter weed spread and population increase is particularly important.

This project has built on the results of previous work on the population biology of exotic woody weeds in northern Australia. It has focused on four species: rubber vine (*Cryptostegia grandiflora*), prickly acacia (*Acacia nilotica*), mesquite (*Prosopis* spp.) and chinee apple (*Ziziphus mauritiana*). The first three of these have been identified as Weeds of National Significance (WONS) under the National Weeds Strategy. The fourth, chinee apple, is also a major problem, most notably in north eastern Queensland.

The efficient use of resources to counter the problem of exotic woody weeds in northern Australia depends on an ability to make effective decisions about what treatments to impose and when to impose them. These decisions must rely on both economic and ecological information.

This project aimed to provide the means to integrate ecological and economic information into decision making for the effective and efficient control and containment of prominent exotic woody weed species in northern Australia. The results are relevant to paddock, property and catchment scales in a variety of land types.

Project Objectives

1. Quantify, for land management purposes (i.e. seasons, plant size), the amount of time that elapses between germination of a woody weed cohort and seed production by that cohort.

2. Determine biologically and economically efficient burning frequencies from the relationship between individual woody weed plant size and density and forage production.

3. Develop management practices (fuel loads, environmental conditions, seasons etc) for the frequency and intensity of burning to control a suite of exotic woody weeds by limiting seed production and direct mortality of juvenile and adult plants.

4. Develop a computer-based decision support package for use by research and extension officers to assist property managers to evaluate ecological and economic priorities for prickly acacia control at the paddock scale.

5. Integrate ecological data (seed production, seedling establishment, growth rates, time to flowering, burning mortality etc) into economic analysis for decision support of rubber vine management at the paddock scale.

SUMMARY OF PROJECT

This project concentrated on developing practices and strategies that are effective in containing and controlling invasive shrubs in northern Australia and based on an understanding of their ecology and population biology. It has built on the work completed during MRC Project CS219. Three of the four species that were the focus of this project are listed among the twenty Weeds of National Significance (WONS) identified in conjunction with the National Weeds Strategy. These are rubber vine (*Cryptostegia grandiflora*), prickly acacia (*Acacia nilotica*) and mesquite (*Prosopis* spp.). The fourth species, chinee apple or Indian jujube (*Ziziphus mauritiana*) is a declared weed in Queensland and the Northern Territory and is present in northern Western Australia. Because each of these species is predominantly a weed of extensive rangelands, there are severe economic constraints to their management. In extensive grazing situations, it is especially important to identify practices and strategies of weed control that are low-cost per unit area of land threatened or infested. Crucial to achieving this goal are questions relating to the timing of weed control activities as well as cost-effective broadacre control techniques.

There have been very significant developments in the management of these species since the early 1990s, in part a result research conducted through NAP Projects. First, it was demonstrated by MRC Project CS219 and other work that fire is very effective against rubber vine, especially small individuals. Mortality rates after a single fire may exceed 90% for plants under 1m high and 50% for plants over 2m high. Second, two biological control agents have become firmly established. These are the now well-known rubber vine rust (*Maravalia cryptostegiae*) and the moth *Euclasta gigantalis*. Both agents repeatedly defoliate rubber vine plants during the growing season. Third, the National Weed Strategy was released in 1997, providing stimuli and mechanisms for encouraging action against weeds and prompting the development of national strategies against specific weeds, most especially against those that were identified as WONS. Each of these developments has been important in relation to MLA Project NAP3.206.

There has been a consolidation of the promising results with prescribed fire against rubber vine. This has involved a continuation of research as well as larger scale demonstrations of fire's effectiveness. Importantly, it has been noted that there are likely to be several important interactions between fire and biological control agents. On the one hand, the defoliating agents are likely to alter the dynamics of herbaceous and litter fuels and so change the frequency and intensity of fires. Fuel loads are likely to be greater where defoliation has increased the availability of light at ground level and fallen rubber vine leaves will add to this, especially in riparian areas. Second, while the effect has not been quantified, repeated defoliation apparently decreases fruit production by rubber vine. This is likely to reduce the rate of spread of rubber vine into new areas (and whatever scale) and the rate of recolonisation of treated areas. Higher probabilities of effective fire and reduced rates of population increase will probably diminish the frequency with which prescribed fires are required and so alter the economic consequences.

This project has:

- Quantified seedling mortality in rubber vine, prickly acacia, mesquite and chinee apple.
- Provided estimates of the time required for prickly acacia to reach reproductive maturity in upland and bore drain habitats.
- Provided estimates of the time required for chinee apple to establish and the time required to reach reproductive maturity.
- Shown that most seedlings of most cohorts of rubber vine and *Prosopis pallida* fail to establish even though their longer-term temporal patterns of recruitment have not been thoroughly quantified.
- Described patterns of grass fuel accumulation in one part of the range of rubber vine and demonstrated that annual burning or fires in successive years are possible in this part of the species' range.
- Demonstrated that two late dry season fires in successive years are sufficient to reduce populations of non-riparian rubber vine to very low levels. The implication is that, thereafter, infrequent burning would be sufficient to keep rubber vine a minor component of the vegetation. Low fire frequency is likely to be adequate for this purpose especially in light of the impacts of defoliating biological control agents on seed production, rates of population increase and, possibly, seedling survival.
- Demonstrated that even four annual late dry season fires have negligible effect on populations of chinee apple. This highlights the need to promote property hygiene as a means of minimising the spread of this species.
- Demonstrated that seedling and juvenile prickly acacia are susceptible to late dry season fires.

- Quantified the responses to *Prosopis pallida* to burning, demonstrating that a single spring fire can kill over 90% of plants. Appropriate grazing management is a prerequisite to burning in western areas where this species of mesquite is prevalent.

- Developed a population model for prickly acacia that can be used to examine the results of particular management scenarios, that encompass, bore drain control, population dynamics, seed dispersal and climatic sequences.

- Refined a model that can be used to investigate the longer-term consequences of using fire or other control techniques in the management of rubber vine. This analysis shows that even infrequent burning can reduce rubber vine populations, and so, presumably impacts. The dynamic situation with rubber vine in northern Australia (notably the impacts of biocontrol agents) makes it difficult to predict the economic consequences of using prescribed fire.

- Incorporated results into the national management strategies for rubber vine and prickly acacia through the activities of the Rubber vine and Prickly Acacia Management Groups.

- Promoted through demonstration and interaction with landcare and other groups the strategic management of exotic shrub weeds on the basis of current ecological knowledge.

OBJECTIVE 1: TIME TO MATURITY

Quantify, for land management purposes (i.e. seasons, plant size), the amount of time that elapses between germination of a woody weed cohort and seed production by that cohort.

Background

Previous studies, notably those reported in MRC Project CS219 have provided quite a comprehensive picture of the demography of four important exotic woody weeds in northern Australia. This work included information of the seed and seedling biology, growth and mortality rates of established plants and seed output. However, at the commencement of this project, there was only a limited amount of quantitative information available on the establishment phase of these important species. An understanding of the dynamics of the establishment phase is critical to the management of these weeds because of its bearing on the timing of management actions. In particular, it is useful to know the typical duration of the phase between germination and seed production as this represents a threshold in the weed invasion process. Ideally the size of a cohort should be significantly reduced before members of that cohort reach reproductive size. A key biological question, then, is, "How long does it take for plants to reach reproductive size?". The corresponding management question is, "How much time is available between germination and reproduction in which to take remedial action?". Work carried out under this project objective was designed to fill in this knowledge gap? Quite clearly, it would be unrealistic to expect there to be a single answer to this question. Significant interspecific variation is likely and significant inter-cohort variation is a certainty as seedlings/juveniles from different cohorts encounter different seasonal conditions.

Establishment of rubber vine and chinee apple

Methodology

This work built on that reported from MRC Project CS219, targeting seedlings of rubber vine and chinee apple at the CSIRO Lansdown Research Station. It monitored cohorts of seedlings of these species that emerged at the study site in the wet seasons of 1996-1997, 1997-1998 and 1998-1999. For each cohort, at least one hundred naturally occurring seedlings of each species were individually tagged within one month of emergence. They were revisited, initially at intervals of less than one month, and then less frequently as mortality rates declined. The fates of seedlings were followed for a up to 920 days. The relationships between seedling age and seedling survival was examined by regression following transformation by:

$$n' = \log_{10}(n+1)$$

where n is the number of seedlings surviving.

Given that no plants have been followed from germination to seed production, the duration of this phase was estimated for chinee apple and rubber vine using data on mortality rates of seedlings and adults and the growth rates of established plants. Estimates of mortality rates for seedlings were compared with those for unburnt, small rubber vine and chinee apple, for which data were available from the experiment described under Objective 3 (see below). The time taken for the mortality rates of seedlings to approach the mortality rates of established plants in height class 1 was calculated on the basis of negative exponential decay functions.

For chinee apple, growth data from the Lansdown experiment was also used to estimate the time taken to reach reproductive maturity. Plants from three height classes were followed for 4½ years. Height increase was approximately linear. Regression and extrapolation were used to estimate the duration of the pre- reproductive phase given that plants do not produce significant amounts of seed until they are 1 m high.

Results

Three cohorts of chinee apple and four cohorts of rubber vine were documented during the study period. Data from these cohorts could be added to those for a single cohort of chinee apple and two cohorts of rubber vine that emerged in 1995.

Cohorts of chinee apple emerged in February 1995, January 1996, February 1997 and January 1998. Cohorts of rubber vine emerged in February 1995, August 1995, January 1996, February 1997, February 1998 and September 1998. At least 100 and up to 498 seedlings of each cohort of each species were tagged (Table 1).

Table 1. Sample sizes at emergence for four cohorts of chinee apple and six cohorts of rubber vine that were monitored at Lansdown Research Station between February 1995 and August 1999.

Cohort (date of emergence)	**No of seedlings tagged**
Chinee apple	
February 1995	111
January 1996	400
February 1997	381
January 1998	498
TOTAL	1390
Rubber vine	
February 1995	118
August 1995	234
January 1996	400
February 1997	410
February 1998	441
September 1998	253
TOTAL	1856

All but one of the rubber vine cohorts that were monitored during the study period declined to extinction within 650 days of emergence. Time to extinction varied between 100 days for the February 1995 cohort and 630 days for the February 1997 cohorts. The only rubber vine cohort of which there were any surviving seedlings at the conclusion of the project was the

September 1998 cohort. There was a single surviving seedling that was 330 days old in August 1999 (Figure 1).

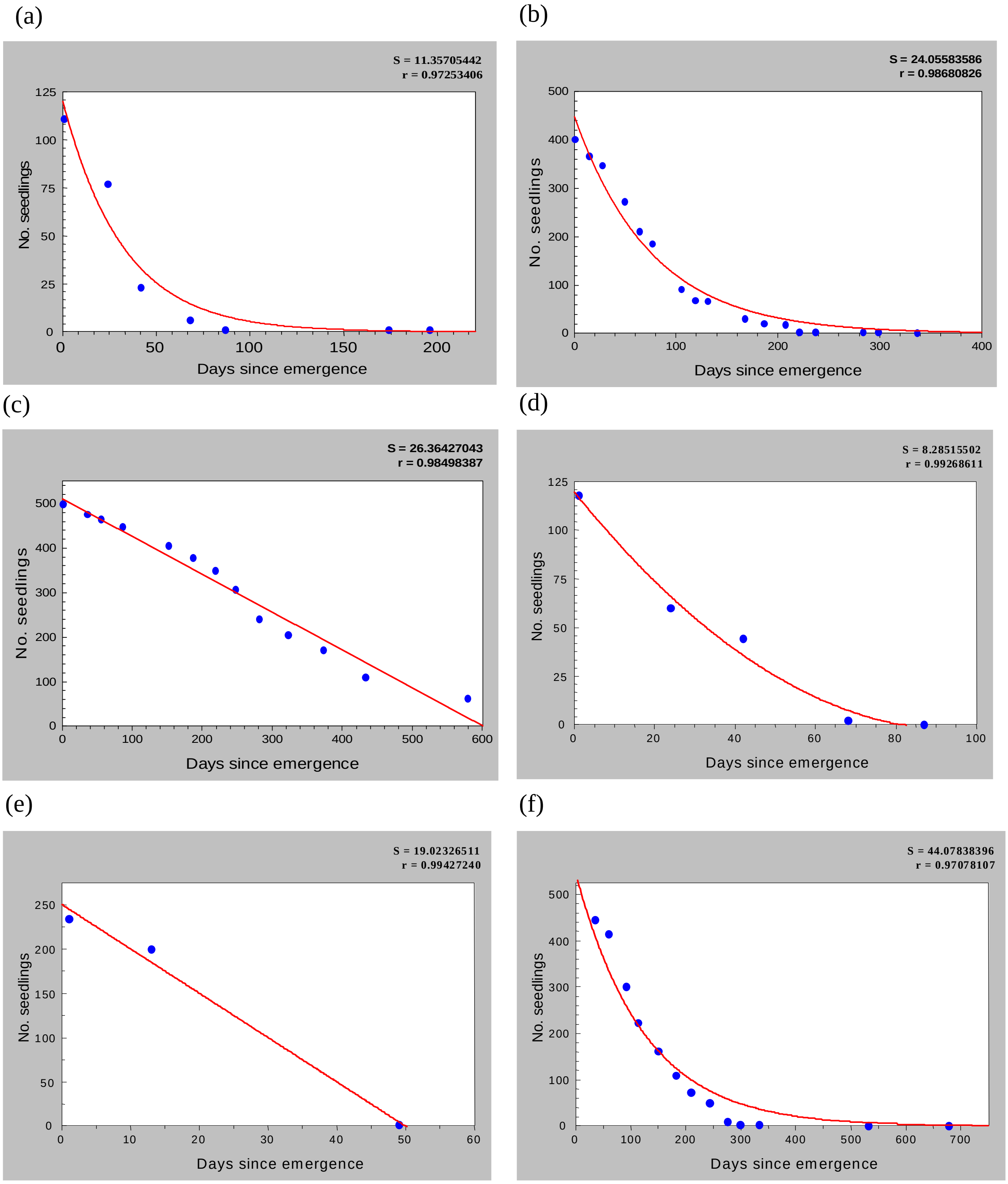


Figure 1. Depletion curves for cohorts of (a) chinee apple, emerged February 1995 (b) chinee apple, January 1996, (c) chinee apple, January 1998, (d) rubber vine, February 1995, (e) rubber vine, August 1995, (f) rubber vine, February 1997, at Lansdown. Fitted curves were either negative exponential or linear.

Two of the chinee apple cohorts (February 1995 and January 1996) declined to extinction within 350 days of emergence. The other two cohorts still had surviving seedlings in August 1999. 12% of the January 1998 cohort were still alive in August 1999, 580 days after emergence. 6.3% of the February 1997 cohort were still alive in August 1999, almost 900 days after emergence.

Data from Lansdown provided estimates of the mortality rates of small but established plants (Table 2). These individuals were all in height class 1 (< 1 m high). For comparison, survivorship of the most long-lived cohorts of chinee apple and rubber vine that were documented were fitted to negative exponential decay functions, namely,

$$y = ae^{bx}$$

The function derived for chinee apple seedlings over a 900 day period was:

$$y = 570.65ae^{-0.00334x}$$

The function derived for rubber vine seedlings over a 700 day period was:

$$y = 542.23ae^{-0.00808x}$$

where, in each case, y stands for the number of seedlings surviving and x is seedling age in days. Differentiation of these equations yields functions that relate mortality rate to seedling age:

$$dy/dx = -abe^{bx}$$

These equations were then used to calculate the age at which seedlings attained mortality rates equivalent to those exhibited by established plants in height class 1. This was done by deriving values for the number of plants dying per day, substituting the value into the differential equation and solving for x. Conversion of the results to years yielded 1.6 and 3.7 for rubber vine and chinee apple respectively. On the basis of this comparison, seedlings that were 2-4 years old can be regarded as established.

Table 2. Mortality rates for small unburned and ungrazed rubber vine and chinee apple at Lansdown Research Station.

	n	**Survival (%) 1994-1998**	**Mortality (%) 1994-1998**	**Annual mortality rate (%)**	**Daily mortality rate (%)**
Chinee apple	62	91.9	8.1	2.0	0.0055
Rubber vine	47	85.1	14.9	3.7	0.01

For chinee apple, height increased in a linear fashion (Figure 2). Growth curves for three different height classes were used to develop a generalised growth curve (Figure 3). Taking into account the mean initial height of 50 cm, and extrapolating to an initial height of zero, it can be estimated by this method that chinee apple would take approximately 6 years to reach reproductive size (ie 1 m) from germination (Figure 3).

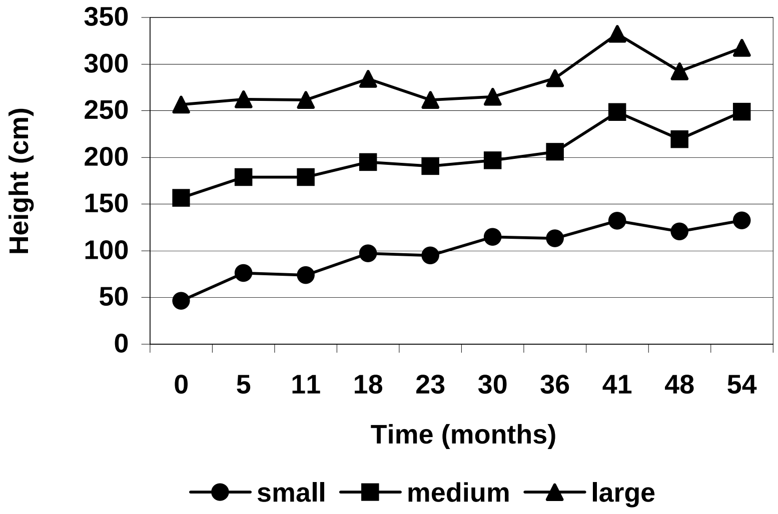


Figure 2. Growth curves for three height classes of chinee apple at Lansdown Research Station. The relationships between plant height (H) and plant age (t) for these three height classes are:

(i) small plants: $H = 1.44\ t + 61.87$ Equation 1
(ii) medium plants: $H = 1.50\ t + 162.18$ Equation 2
(iii) large plants: $H = 1.11\ t + 252.17$ Equation 3

Time zero for the three height classes are times when height = 50, 150 and 250cm respectively. Approximate time to reproductive size (H = 100cm) was calculated using Equation 1:

$H = 1.44\ t + 61.87$

when $H = 100$, $100 = 1.44\ t + 61.87$, $t = 26.48$ months

when $H = 0$, $0 = 1.44\ t + 61.87$, $t = -42.96$ months

Therefore, time to reach H = 100cm = 26.48 + 42.96 = 69.45 months = 5.79 years

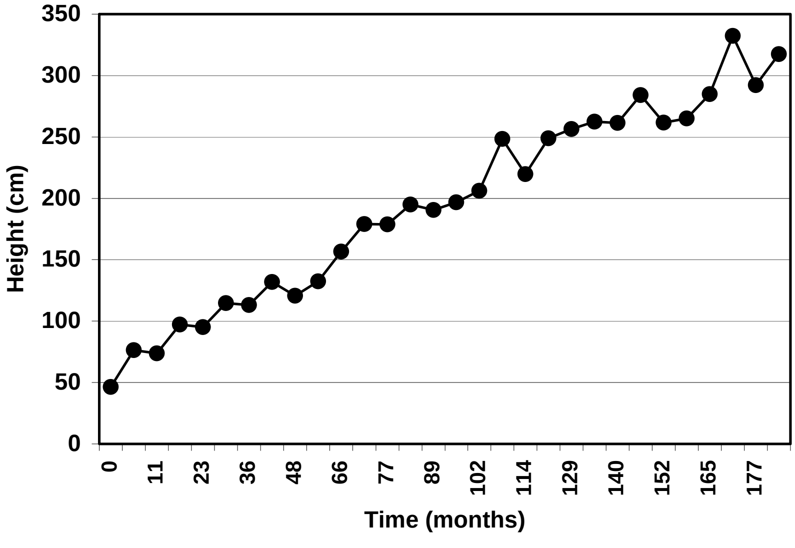


Figure 3. Generalised growth curve for three height classes of chinee apple. In this diagram, time zero corresponds to the time at which plants were *ca.* 50 cm high.

Establishment of mesquite

Methodology

Seedling emergence, establishment, survival and growth rates were investigated in a long-term field experiment to improve our understanding of the invasion dynamics of *P. pallida,* the most widespread of the mesquite species growing in Queensland. The study site was located on a floodplain near Hughenden, where an infestation of *P. pallida* covers an estimated 20,000ha. Long term annual mean rainfall for this region is 482 mm. Approximately 80% of this falls between October and March, which is defined here as the wet season.

The experiment commenced in October 1995, following the removal of domestic stock from the site. Seedling emergence was followed in both undisturbed areas and in areas where all *P. pallida* plants had been removed. The objective of following seedling emergence in cleared areas was to understand the rundown of the seed bank in areas where nil replenishment occurs. For the undisturbed sites, seven areas with varying densities of *P. pallida* were selected, and permanent transects 60 m long by 10 m wide were established within each area. Monthly recordings were undertaken between October 1995 and August 1998 to identify emerging *P. pallida* seedlings and monitor their subsequent survival and growth (stem basal area and plant height). Each seedling found was also checked to determine if it had emerged from dung. The number of reproductive *P. pallida* trees in each transect was recorded every month during the 1997/98 wet season. Trees with all or part of their base within the transect were assessed for the presence of pods or flowers. If either was detected, the tree was classed as 'reproductive'. For cleared sites, four areas were chosen, with all trees bulldozed out in October 1995. A similar approach to that used in undisturbed areas was then adopted to measure seedling emergence. Climatic conditions during the study period were measured using an automatic weather station.

Results

More than 12,000 seedlings emerged in the seven undisturbed transects during the three wet seasons encompassed in the study period. Eleven seedling cohorts were identified according to the distinct rainfall events on which they germinated. There were three cohorts in both 1995/96 and 1996/97, and five cohorts in 1997/98. Rainfall for each of the three wet seasons (Figure 4) was less than the long term mean (396 mm), averaging 349, 370 and 310 mm for the1995/96, 1996/97 and 1997/98 seasons respectively.

A highly significant positive relationship ($R^2 = 0.762$, $P<0.01$) occurred between the density of reproductive trees and the number of seedlings which emerged per unit area (Figure 5). Seedling emergence varied from 1183 seedlings per ha during the 1995/96 wet season in an infestation density of 67 reproductive trees per ha, to 58 468 seedlings per ha during the 1997/98 wet season in an infestation density of 1283 reproductive trees per ha.

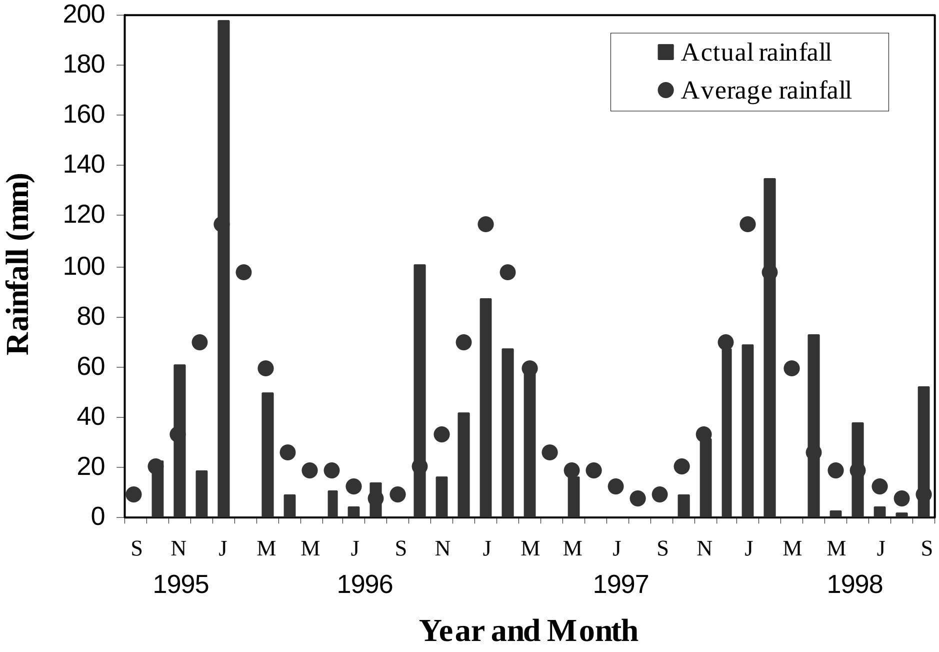


Figure 4. Actual monthly totals and long-term monthly mean rainfall for Hughenden (ν actual rainfall; λ average rainfall).

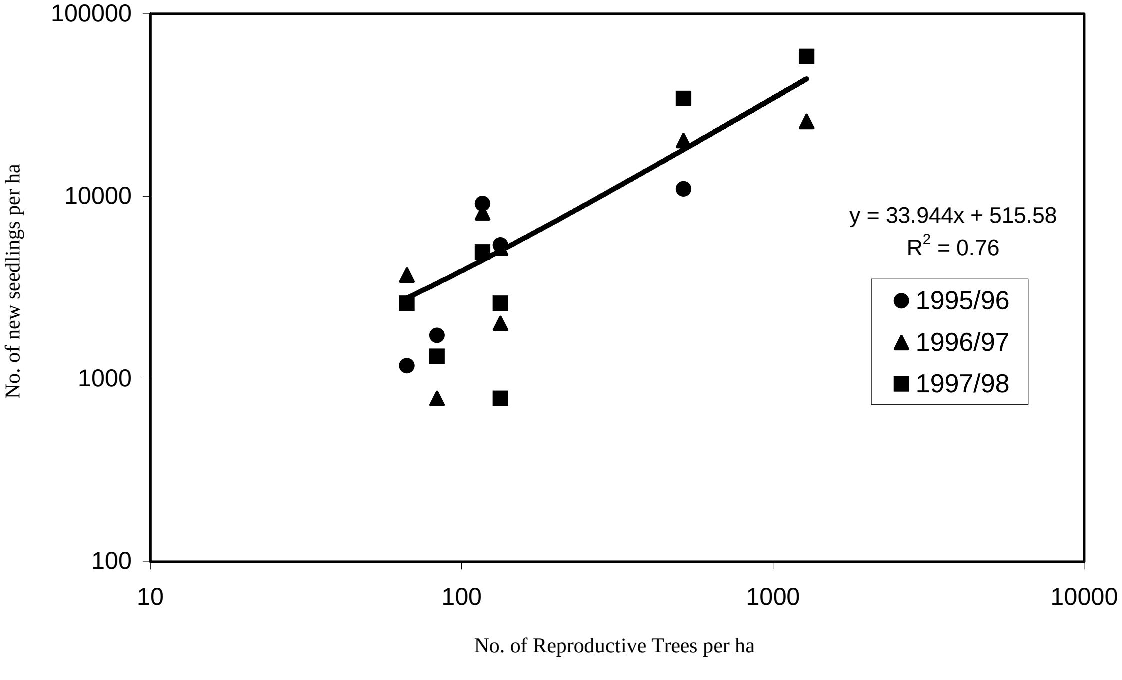


Figure 5. Relationship between reproductive plant density and seedling emergence.

In the undisturbed transects, seedling emergence increased greatly over the three wet seasons, from an average of 4388 seedlings per ha in 1995/96, to 9391 in 1996/97 and 12362 in 1997/98. Of the total 12300 seedlings that emerged during the study, only 3.7% survived through to August 1998, of which 2.3, 0.4 and 1% were 1995/96, 1996/97 and 1997/98 seedlings respectively. Survival of cohorts of *P. pallida* after 12 months averaged 4% and ranged from no seedlings surviving to as many as 16% remaining alive (Figure 6). Cohorts that exhibited high survival tended to receive high initial rainfall on which germination occurred and good follow up rainfall soon after germination.

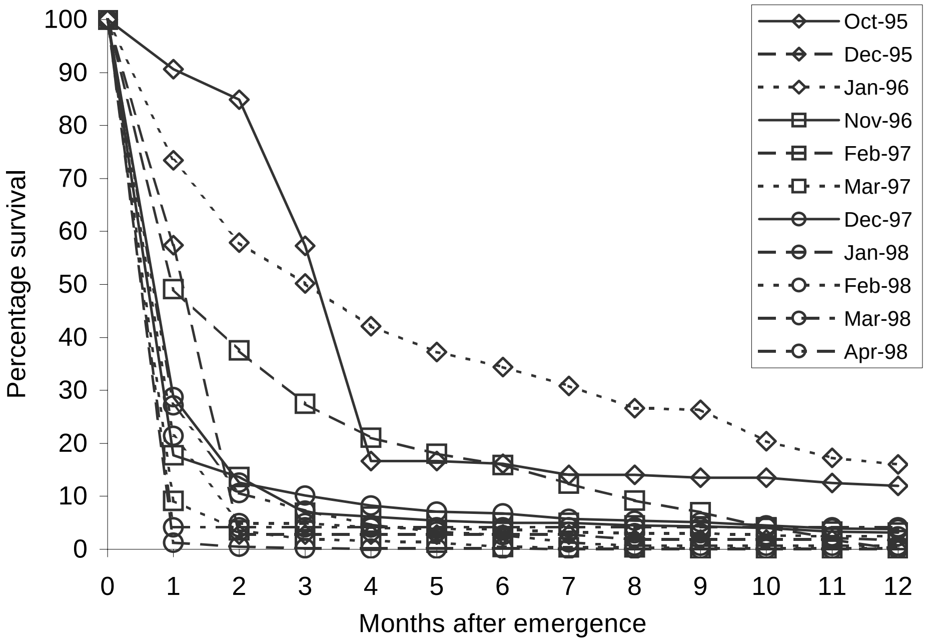


Figure 6. Twelve month survivorship curves for 11 cohorts of *P. pallida* that emerged between October 1995 and April 1998.

In the cleared plots, where all reproductive trees were removed to prevent seed bank replenishment, there was no significant decrease in emergence over the following three wet seasons (Figure 7). In the absence of domestic grazing there are two possible explanations. Firstly, that dormancy mechanisms are controlling the availability of seed to germinate, leading to a staggered pattern of germination and emergence and, secondly, that native and feral animals transported seeds into the clean areas. It is most likely that both factors were contributing in this situation.

During the life of the project only one of the 12300 seedlings recorded reached reproductive maturity. The plant was 3 years old at the time and was from the January 1996 cohort. It had been growing in a more favourable location than most seedlings; in one of the cleared plots and situated in a low lying area where rainfall tended to pool. It had a height of 3.0 m and an average basal diameter of 5.6 cm when it reached maturity. Consequently, it can be fairly safely concluded that under average rainfall conditions it would probably take *P. pallida* longer than 3 years to reach reproductive maturity.

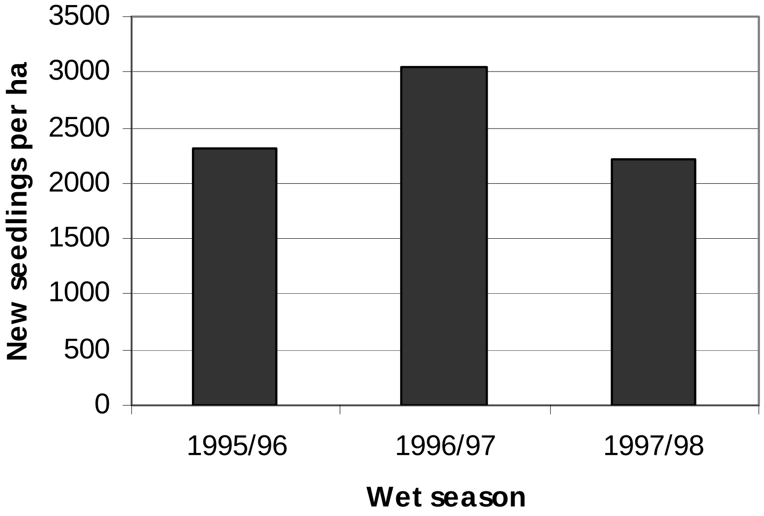


Figure 7. Seedling emergence over three wet seasons following the removal of all mesquite plants.

Establishment of prickly acacia

Methodology

This work was carried out in the Mitchell grass plains between Hughenden and Julia Creek. This area has been identified as an area of high infestation (Carter 1989, Mackey 1997). Permanent sites were established at Nicoleche ($20^0$54.03'S $144^0$10.09'E) and Marathon ($20^0$53.45'S $143^0$33.96'E and PT $20^0$53.20'S $143^0$41.08'E) stations which are cattle properties, and Allaru ($20^0$46.66'S $143^0$09.24'E) and Garomna ($20^0$40.39'S $141^0$47.04'E and $20^0$42.24'S $141^0$51.13'E) stations which run sheep.

Two habitat types were sampled, uplands which were non-riparian Mitchell Grass downs where there was no surface water, and artesian bore drains where there was permanent surface water. Tagged adult and juvenile (non-reproductive) plants were measured for size and density in April each year to record annual growth at Allaru. Plants were measured for height, canopy width in two directions, diameter of the trunk at 30cm, browse height (height from the ground at which the canopy begins). This allowed the canopy of each tree to be approximately estimated. Mortality of both adult and juvenile plants at all sites was monitored on an annual basis. The density of adult and juvenile populations were estimated at each site.

Seedlings were chosen randomly from each of the sub-sites. This allowed a range of micro habitats within each sub-site to be sampled. Sixty six seedlings were individually tagged at every sub-site where there were enough to do so. Seedlings were tagged immediately after access was possible at the sites following the wet season to allow sampling of populations at their maximum. Further sampling took place at approximately six weekly intervals. The following measurements were taken of each seedling: total height; width of canopy; number of leaves (not including cotyledon); distance to nearest adult and juvenile acacia; nearest living plant of other species; whether the seedling was directly under canopy of nearby adult plants; and whether seedling had spines. Seedling mortality was recorded on a six-weekly basis. Seedling density was estimated by recording the distance from random points at each field site, to the nearest seedlings in 4 quadrants. Number of seedlings per clump was recorded.

In order to extrapolate growth rates for plants of differing sizes, annual growth of monitored plants was regressed against height. These growth rates were used to extrapolate over a 30 year period starting from the average size of seedlings when first recorded after germination in their first year. Factorial analysis of variance (ANOVA) was used to investigate the significance of seedling attributes in determining survival. Average seedling, juvenile and adult mortality during this study was used to calculate reproductive plant recruitment from seedling densities.

Results

Growth rates vary among plants of different sizes and between bore drain and upland habitats. In bore drain habitats seedlings and juveniles have higher average growth rates (from 25 to 50 cm year^{-1}) than large trees (0 to 10 cm year^{-1}), with the largest trees (>6 m high) actually having negative growth during the three years of this study (Figure 8a). In contrast, plants in upland habitats had low and constant rates of growth (ca. 5 cm year^{-1}) from very small seedlings through to adult (Figure 8b).

If we extrapolate growth rates seen in the three years of this study to a hypothetical seedling cohort, bore drain plants grow much more rapidly than upland plants (Figure 9). The bore drain cohort initially grew rapidly (up to 250 cm in 8 years) and then slowed as plants got bigger (22 years from 250 cm to 500 cm). The upland cohort grew at a constant rate and plants grew an average of 200 cm in 30 years.

Mean size of plants at first flowering was 255 cm in monitored populations during this study. At growth rates seen during this study it would take approximately 8 years for plants to reach podding point, compared to 37 years in upland populations.

There was no direct relationship between total wet season rainfall and *Acacia nilotica* growth rates (Table 3). Despite low rainfall in 1995/96 and 1997/98 wet seasons highest average growth rates were observed in the former period for upland trees and the latter period for bore drain trees. Conversely, growth rates were only medium to low in 1996/97 when rainfall was much higher.

First year seedling mortality was high in all three monitoring periods (Figure 10), varying from 98 to 99%. Percentage mortality was high in both wet (1996/97) and medium to dry years (1995/96 and 1997/98); in fact highest mortality was seen in the wettest year. No consistent difference in seedling mortality was seen between habitats.

Initial seedling size and distance to the nearest *Acacia nilotica* seedling were found to be the most significant determinants of early seedling survival (up till 50% mortality) during this study (Table 4). In contrast, shading by tree canopy, distance to adult stem, distance from nearest herbaceous vegetation and habitat type, did not significantly influence seedling survival in the first year after germination.

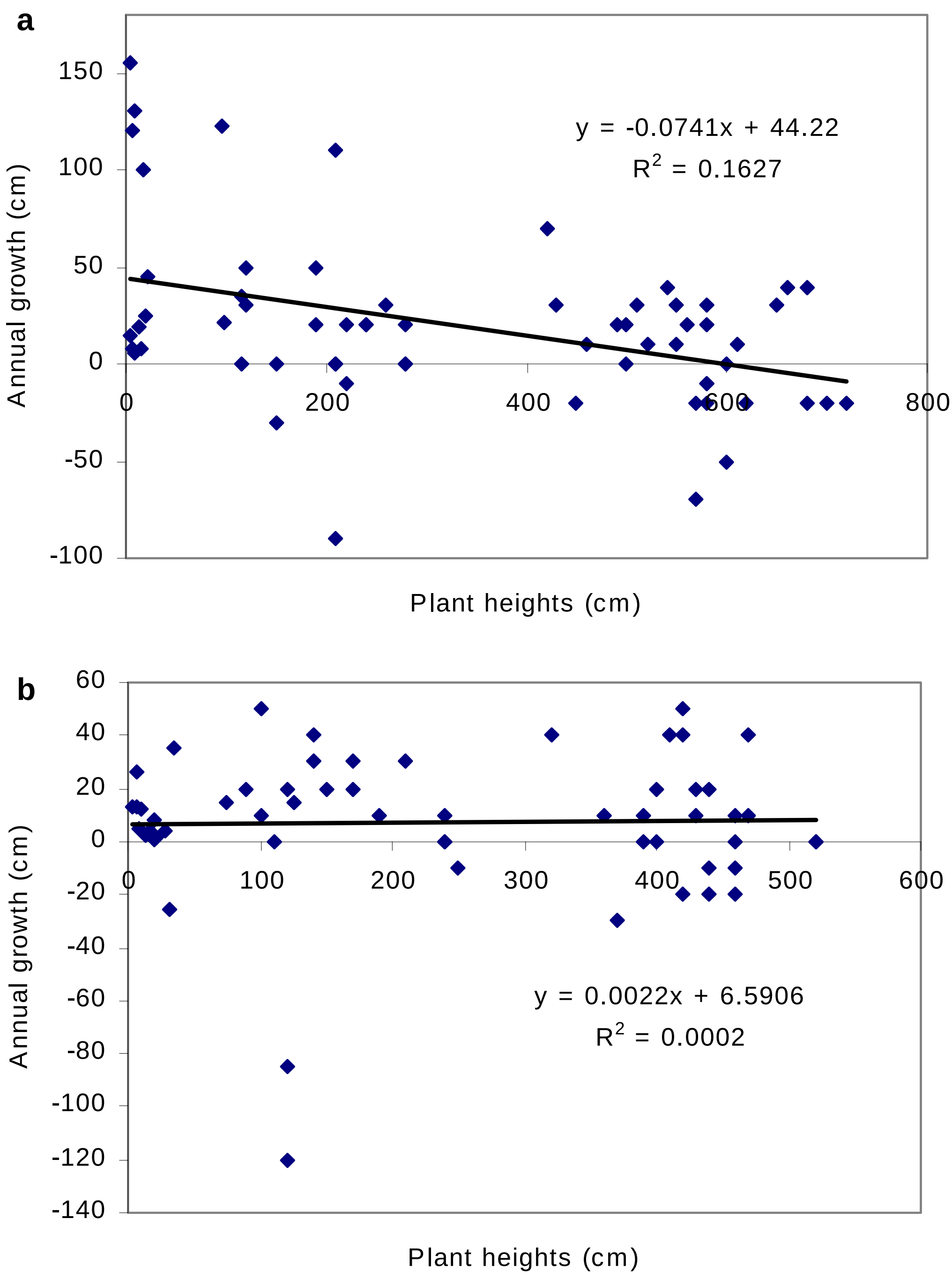


Figure 8. Annual plant growth for plants of differing heights in a) bore drain and b) upland habitats. Regression lines for the relationships between plant height and annual growth are shown with R^2 values and predictive equations.

Annual mortality during this study was highest for seedlings and much lower for juvenile (>1 year) and adult (reproductive) plants (Table 5). Based on growth rates of 8 years to the first pod production (see Figure 9), recruitment is likely to be of medium, low and low density (as per March 1995) following seedling germination recorded after a wet year (1996/97).

Table 3. Mean (s.e.) growth of *Acacia nilotica* in bore drain and upland habitats in three years with different rainfall.

Wet season	Rainfall (mm)	Bore drain tree growth (cm/yr)	Upland tree growth (cm/yr)
1995/96	352	7.7 (5.9)	18.0 (7.6)
1996/97	537	13.4 (9.4)	2.5 (7.0)
1997/98	284	33.0 (10.2)	7.5 (4.0)

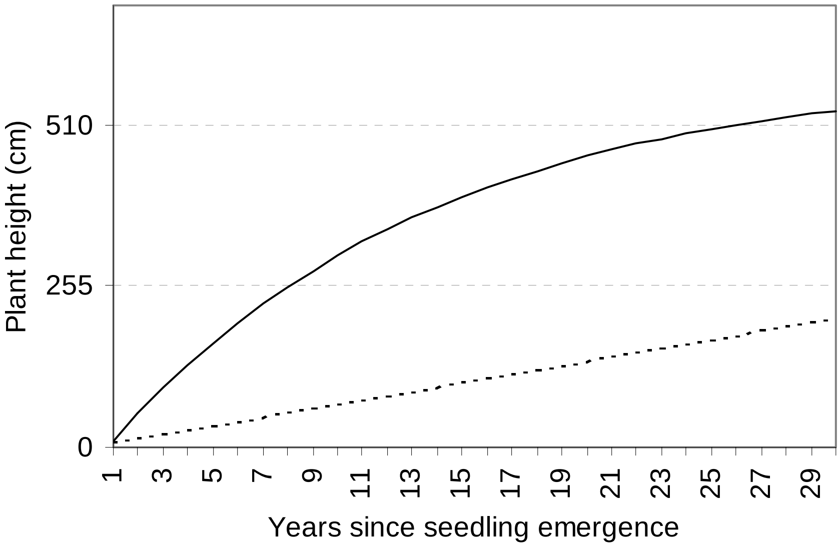


Figure 9. Interpolated growth rate of *Acacia nilotica* in bore drain (_____) and upland (------) habitats. The lower horizontal line represents the mean height of plants at first flowering.

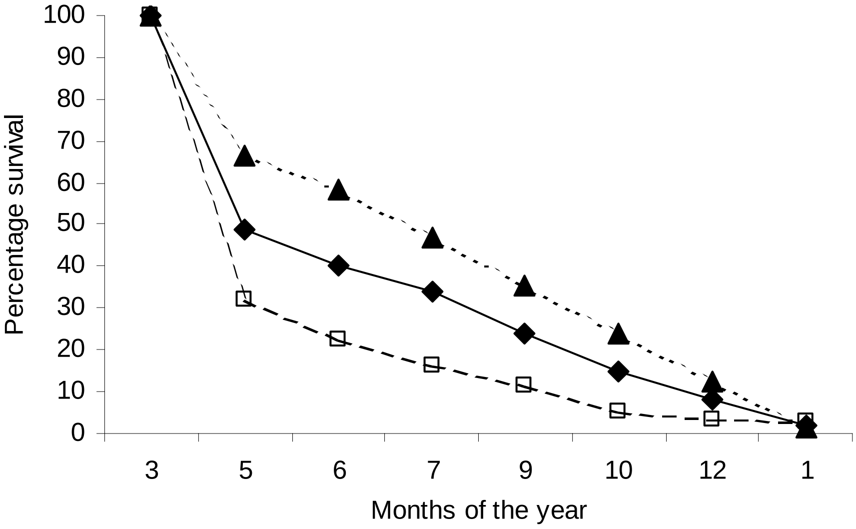


Figure 10. Seedling survival from March through to the following January in the following year in 1996 (♦), 1997 (□) and 1998 (▲).

Table 4. Analysis of Variance of seedling attributes and position which influence mortality up until >50% have died.

Seedling attribute at emergence	50% seedling survival 1996	50% seedling survival 1997
Height	0.007 **	# <0.001 ***
Canopy width	0.003 **	<0.001 ***
Leaf number	0.010 *	# <0.001 ***
Under adult tree canopy or not	0.088 n.s.	0.445 n.s.
Distance from nearest adult trunk	0.910 n.s.	0.150 n.s.
Distance from nearest seedling	# 0.021 *	# <0.001 ***
Distance from other vegetation	0.213 n.s.	0.986 n.s.
Habitat type	#	0.125 n.s.

*** highly significant ($P<0.001$), ** significant ($0.001<P<0.01$), significant ($P<0.05$), # heterogenous data ($\log_e$ transformed).

Table 5. Mean annual mortality for monitored seedling, juvenile and adult *Acacia nilotica*.

	Seedlings	Juveniles	Adults
Mean annual mortality (%)	98.6	13.8	6.0
Recruitment scenario* (8 years)			
Dense: Bore Drain (ha^{-1})	20,000	280	115
Medium: Riparian (ha^{-1})	1,940	27	10
Sparse: Upland (ha^{-1})	590	7	3

Based on seedling emergence after good rainfall in the wet season of 1997/98

Conclusions

From the work described here we have been able to make estimates of the time taken for rubber vine and chinee apple to grow from germination to seed production. Such estimates should be used only to present a general picture of the population dynamics of these species. This is because there will always be enormous variation in the growth rates of any particular species. The growth rates will be determined by the resources available to the plant especially water and nutrients. Thus the time taken to reach the reproductive phase will vary between climatic zones, with soil types, seasonal conditions and landscape position. For chinee apple and rubber vine it appears that, under field conditions, time from germination to maturity is of the order of 5-10 years. This is in the relatively high rainfall zone of Lansdown Research Station, though the time during which the data were collected did include several below average years (Figure 11). More reliable measures of the time to reproductive maturity must be based on monitoring of plants from the times of germination to establishment and hence to first reproduction.

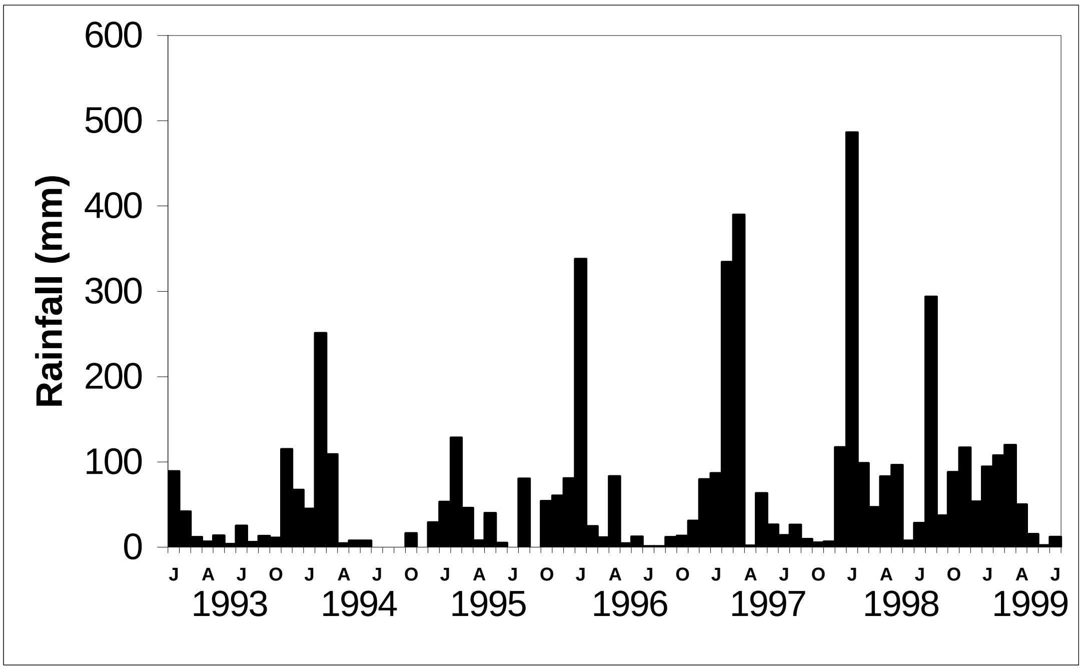


Figure 11. Monthly rainfall totals for Lansdown research station for the period January 1993 until July 1999.

The ability of *P. pallida* to have at least three germination events every year for three consecutive years of below-average wet season rainfall provides an indication of the invasive potential of this plant. Even with an average survival rate after 12 months of 4%, reinvasion of cleared areas or thickening up of scattered infestations can take place. However it will be the above average rainfall years when survival is higher that the rate of invasion will markedly increase.

The shortest time that *P. pallida* took to reach reproductive potential in the field was 30 months. Consequently, land holders should have sufficient time to implement control programs to treat regrowth before replenishment of the seed bank will occur.

According to growth rates projected for hypothetical bore drain and upland cohorts, *Acacia nilotica* plants may take between 8 years and 37 years to produce their first pods and from 30 to 70 years to produce large adult trees (5 m tall) typical of populations seen today. Projected growth rates for the bore drain cohort appears to be closer to growth seen in real *Acacia nilotica* populations in mitchell grasslands. Historical records of population growth and expansion indicate that most upland areas have only been invaded by this species since the 1970’s when a series of wet years, in combination with replacement of sheep by cattle, allowed trees to become established here (Thompson 1992, Mackey 1998). Aerial photography in mitchell grass landscapes between 1960 and 1992 indicates that the majority of upland areas remained free of adult trees until after 1974 (Brown and Carter 1998). This historical evidence indicates that most trees in the mitchell grasslands today are about 26 years old. Growth rates as slow as those seen in upland trees during this study could not have produced extant populations of *Acacia nilotica*.

It should be emphasised here that individual plant growth rates can be far greater than the average seen in this study. Two plants monitored from germination at another site took only

18 months to produce their first flowers (Radford et al. unpublished). This shows that in ideal conditions very rapid cohort maturity can occur.

The low growth rates and mortality observed in upland populations during this study indicate that *Acacia nilotica* populations are currently in a period of decline. At growth rates observed a cohort originating now would take approximately 30 years to produce pods (Figure 9). At the mortality rates observed in upland populations at present (Table 5) none of these plants would survive to produce seeds. This decline has apparently been evident for some years with reports of dramatic decreases in upland tree density and significant die-back occurring during the predominantly dry decades of the 1980' and 1990's. This data suggests that most of the upland populations of today were recruited during a period of more rapid growth and lower mortality than seen at present. Perhaps in the wet years of the early 1970's growth rates in uplands were similar to that seen in bore drain plants in this study.

The growth rates recorded here are only for the survivors. It is the mortality rate combined with seedling density that determines the severity of *Acacia nilotica* infestations.

Mortality recorded for seedlings, juveniles and adult trees in this study suggest that recruitment resulting from germination events would result in medium to sparse populations (as per March 1995) depending on starting seedling density. Such recruitment events could only replace existing populations, not leading to increased population density, based on adult tree mortality rates. Mortality rates recorded in this study may not be typical of major recruitment events such as what occurred in the 1970's.

A major factor limiting recruitment potential of *Acacia nilotica* during the period of study was the small proportion of seeds that germinated. Although seedling germination was a maximum of 590 seedlings/ha in upland habitats in this study, up to 15,000 seeds/ha were deposited in upland areas in cattle dung in 1997 (Radford et al. in prep.). Previous studies have shown that *Acacia nilotica* seeds can continue to germinate up to 400 days after wetted. Such very high seed germination may only occur after extended flooding as seen in 1973 in many areas of the mitchell grasslands. During this period soils were wet for extended periods. If 60% of *Acacia nilotica* seeds in dung are viable (Radford et al. in prep) as many as 9000 seeds/ha may germinate. In such circumstances even high mortality rates of seedlings will still result in a significant increase in tree density in upland areas.

Window for management of **Acacia nilotica**

The crucial period for predicting future population increase is soon after seedlings have germinated. This is because options for control of seedlings are cheaper and more assured of success than for either juveniles or adult trees, and because this strategy prevents most plants producing seeds which could lead to future population increase. As indicated above, seedlings are much more vulnerable to environmental factors than larger plants. This is shown by their much higher mortality rates. Seedlings have been shown to have low tolerance to heavy browsing and trampling by sheep and goats (Pratt and Knight 1971, Cobon 1994) and are vulnerable in dense patches to foliar herbicides (March 1995). Seedlings have also been found to be sensitive to fire in the first and second years after germination. Fire has the advantage of being applicable over large areas.

If seedlings occur at high density towards the end of their first dry season (>50/ha) measures should be taken to increase their mortality using selective browsing, foliar sprays or fire.

Such measures will have the advantage of preventing future population increase before expensive control becomes required. Reactive control of *Acacia nilotica* seedlings requires early detection. High seed germination occurs in years of above average rainfall or after large rainfall events.

A major concern with regards to future *Acacia nilotica* infestations is the large numbers of juvenile plants presently found in many areas of the mitchell grasslands. Juvenile plant density (plants less than 255 cm) at experimental sites was equivalent to density of adult plants (means between 16 and 43 plants/ha). Many of these juvenile plants could produce seeds in less than 5 years. This means that the density of seed producing *Acacia nilotica* plants may double in only a short period if control is not implemented and particularly if a few years of good rainfall occur. Unfortunately, juveniles are much more robust than seedlings and require conventional control methods (as per March 1995) for significant mortality to occur.

OBJECTIVE 2: BURNING FREQUENCIES

Determine biologically and economically efficient burning frequencies from the relationship between individual woody weed plant size and density and forage production.

Background

While it is widely accepted that woody vegetation reduces herbaceous production, there are few data available for examining the relationship between woody plant density and herbage yield. These relationships have most commonly been described by measuring herbage responses to the removal of woody plants or by examining herbaceous biomass at sites that vary in terms of the density or biomass of trees and shrubs. These scenarios are somewhat different from a situation in which exotic woody species are invading or increasing in density. In particular, the economics of control of woody weeds will be critically influenced by the shape of the relationship between increasing weed density or biomass and decreasing herbage yield. Some aspects of these relationship were explored by Brown and Ash (1996). They argued that an “ecological threshold” for a woody weed invasion preceded an “economic threshold” for woody weed control by many years. In other words, the economic impacts of woody weed invasion are not felt until long after the invasion process has become ecologically inevitable.

Where fire is a woody weed control option, notably for rubber vine and *Prosopis pallida*, the relationship between herbage yield and woody weed density will have a bearing on how frequently it will be necessary to apply prescribed fire. Though prescribed burning may be more cost effective than some other methods of control, there are some obvious costs involved. These stem largely from the need to construct and maintain effective firebreaks but, perhaps more importantly, from animal production foregone while paddocks are destocked to allow fuel accumulation or pasture recovery. Because of this it is desirable to use fire only as often as necessary to prevent woody weed increase.

The experimental work carried out under this objective was designed to quantify the relationships between woody weed abundance and forage production.

Herbage yield as a function of fire-induced mortality

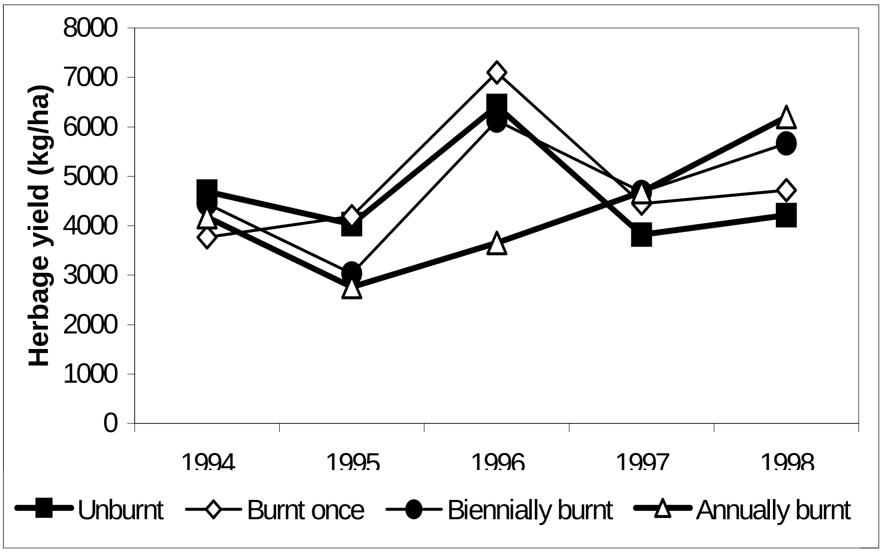
Methodology

The relationship between herbage production, the abundance of woody weeds and fire frequency was examined at Lansdown Research Station using large plots subjected to different fire frequencies. As outlined in the material presented under Objective 3 (below), plots were either unburned, or burned once, twice biennially, or four times annually (Figure 17). These plots carried rubber vine and chinee apple at densities of 210 plants/ha and 689 plants/ha respectively. Herbaceous biomass in each of the nine plots was estimated annually during the mid-late dry season, prior to burning, using the BOTANAL sampling procedure. Herbaceous biomass was examined as a function of annual rainfall totals and percentage rubber vine decline as a result of prescribed burning.

Results

At the start of the experiment, pasture yields of all plots were in the range 3713-4843 kg/ha. Over the course of the experiment, pasture yield on unburnt plots ranged from 3820±60 kg/ha in 1997 to 6416±626 kg/ha in 1996. There were no obvious trends in pasture yield over the course of the experiment for any of the burning treatments (Figure 12). Most plots had higher pasture yields in 1996 than in 1995 or 1997 but some plots also had high yields in 1998.

(a)



(b)

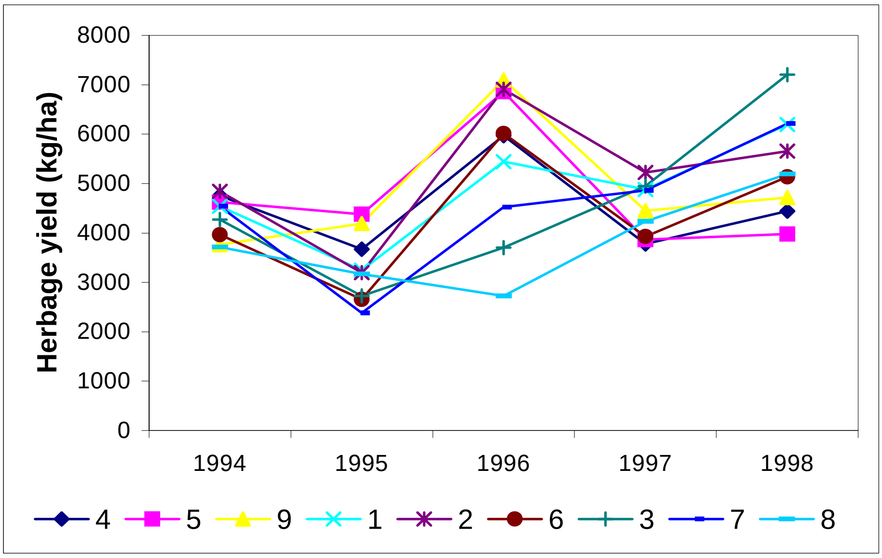


Figure 12. Herbage yield (kg/ha) in the mid-late dry season over five years for four burning regimes at Lansdown Research Station. (a) Herbage yield on individual plots, ordered according to treatment. (b) Mean herbage yields for unburnt, once burnt biennially burnt and annually burnt treatments.

Herbage yield and composition in the vicinity of individual shrubs

Methodology

Herbage yield and composition were examined in the vicinity of individual shrubs at Lansdown Research Station. Ten established plants each of rubber vine and chinee apple were selected and fenced to preclude livestock. Herbage yield and composition beneath and away from the canopy of each individual shrub were estimated at the end of the growing season (May) using the BOTANAL technique.

Results

Patterns of species yield and composition were erratic. For a large proportion (13 out of 20) of the individual shrubs sampled, biomass in the open (away from canopy) exceeded that beneath the canopy. For the remainder, herbage yield was similar beneath and away from the canopy except in one case, where yield beneath the canopy exceeded that in the open. For two individual shrubs, the herbage yield beneath the canopy was zero. The mean yield away from shrub canopies was almost twice that beneath canopies (Figure 13).

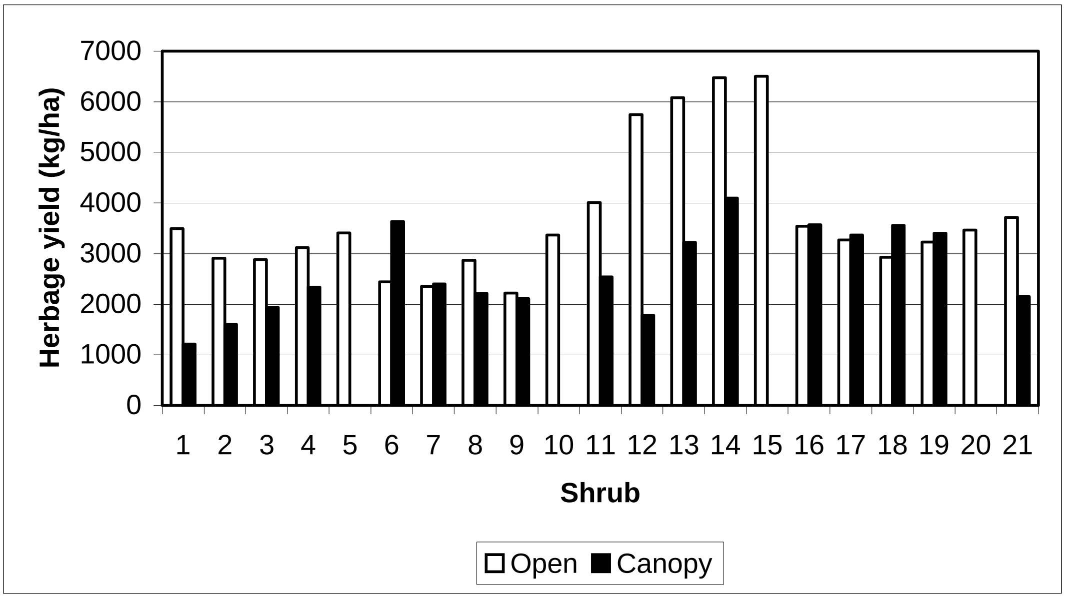
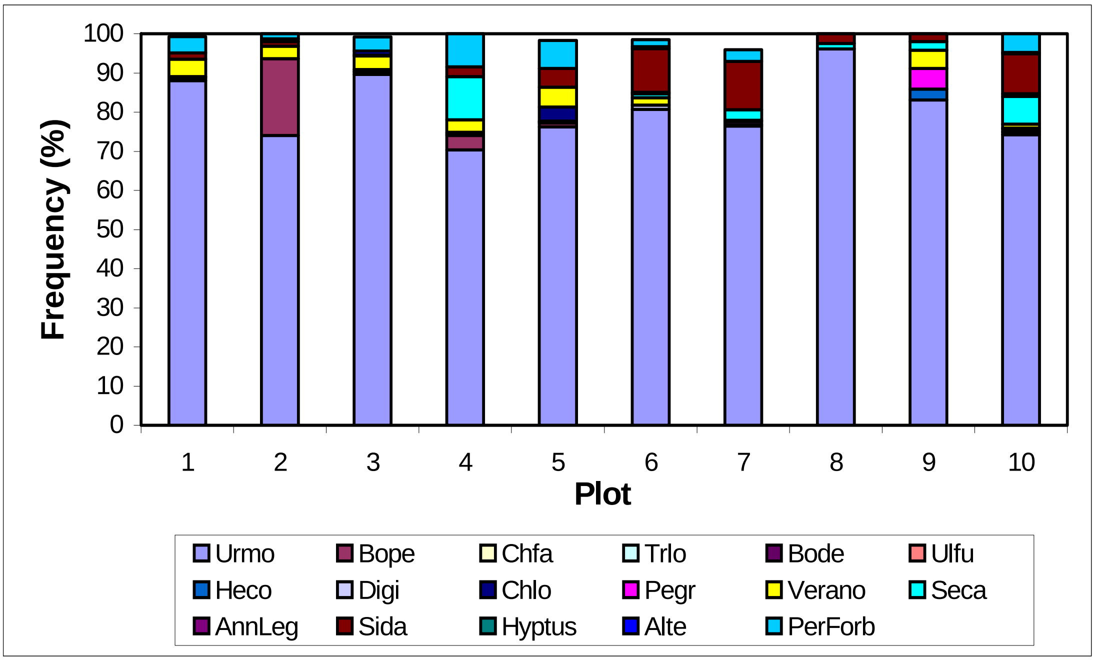


Figure 13. Herbage yield (kg/ha) beneath (canopy) and away from (open) individual shrubs of rubber vine (shrubs 1-10) and chinee apple (shrubs 11-20). Mean yields are recorded as shrub 21.

Herbage composition was also highly variable between sampling sites (Figure 14). Pastures beneath and away from shrub canopies were both dominated by the exotic perennial grass *Urochloa mosambicensis*. The introduced pasture legumes were more abundant away from shrub canopies while the weedy forbs, particularly *Alternanthera* and *Sida* were more abundant beneath canopies. The forb:grass ratios of the pastures did not differ between open and canopy locations (Figure 15).

(a)



(b)

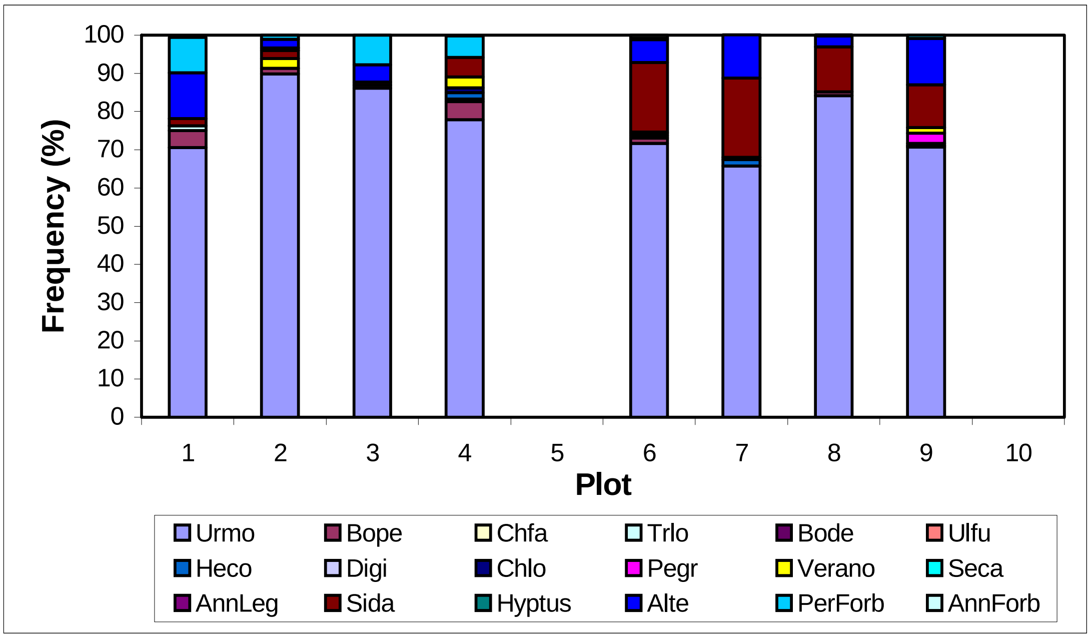


Figure 14. Species composition (frequency %) in (a) open plots and (b) beneath canopies of rubber vine shrubs. Species codes: Urmo *Urochloa mosambicensis*, Bope Bothriochloa pertusa, Chfa *Chrysopogon fallax*, Trlo, Bode *Bothriochloa decipiens*, Ulfu, Heco *Heteropogon contortus*, Digi *Digitaria* spp., Chlo *Chloris* spp., Pergr Other perennial grasses, AnnLeg Other annual legumes, Sida *Sida* spp., Hyptis *Hyptis suaveoloens*, Alte *Alternanthera* sp., PerForb Other perennial forbs, AnnForb Other Annual Forbs.

Conclusions

Herbage yields on plots at Lansdown Research Station were sufficient for annually burning. This was true even in years when total annual rainfall was well below average. For instance, the annual rainfall totals for 1993-1996 were 405, 471, 557 and 613mm respectively whereas the average annual rainfall is 861mm (Figure 16). Even on annually burnt plots, average herbage yields exceeded 2500 kg/ha every year. Given reasonable fuel continuity, this fuel load is adequate for a fire that would be effective against rubber vine (see next section). These data indicate that in the relatively mesic environment of Lansdown, annual burning is generally possible. However, the results in the next section indicate that annual burning is not necessary for effective management of rubber vine.

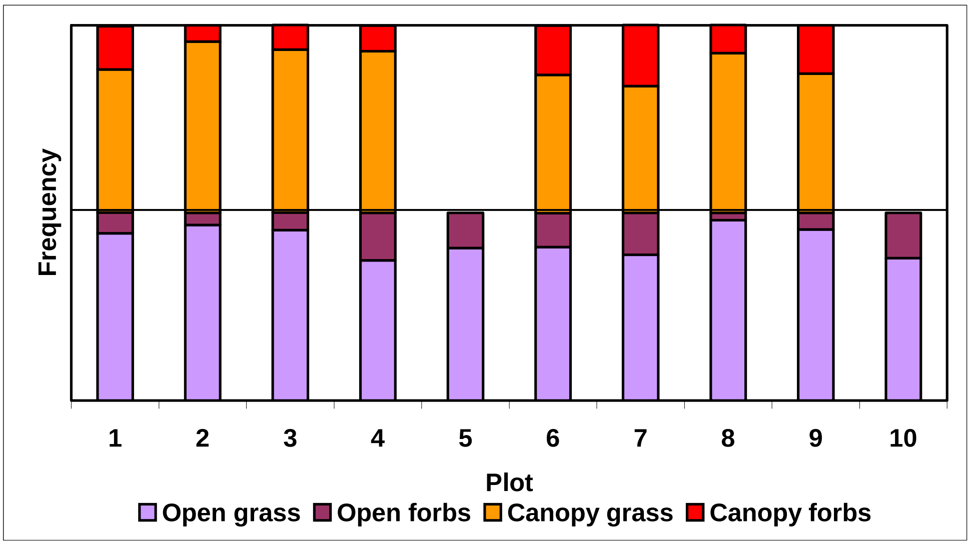


Figure 15. Relative frequency of grasses and forbs beneath the canopies of ten rubber vine shrubs and on adjacent open plots.

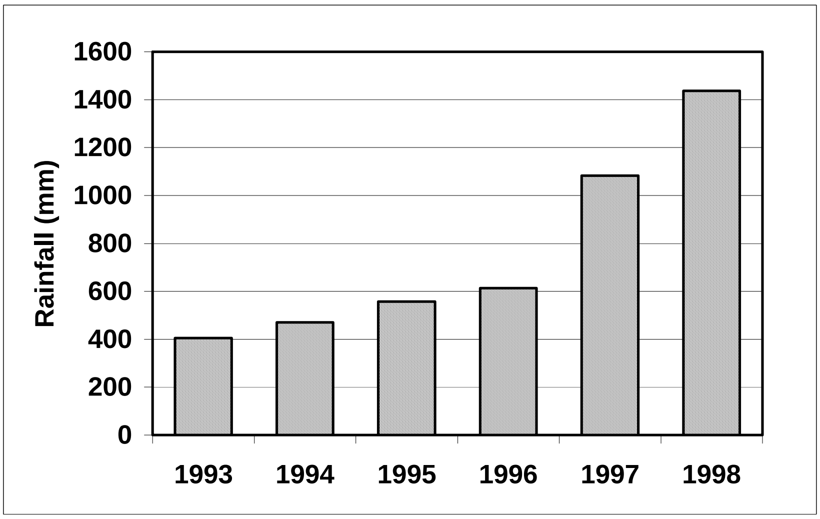


Figure 16. Annual rainfall totals (mm) at Lansdown Research Station between 1993 and 1998. The average annual rainfall at Lansdown Research Station is 861mm.

The patterns of pasture species composition and yield in the vicinity of individual chinee apple and rubber vine shrubs were erratic. There is great variation between individual shrubs in the pasture responses, probably due at least in part to the variation in shrub morphology. The yield of herbage beneath individual shrubs whose canopies reach the soil surface was apparently lower than that beneath shrubs whose canopies were not so structured. The environment beneath shrub canopies appeared to favour weedy forbs at the expense of perennial grasses. Other factors that may contribute are the arrangement of shrubs relative to one another and seasonal conditions. The complexity of these interaction make it difficult to quantify the effects of shrubs on pasture yield and composition.

OBJECTIVE 3: FIRE IN WEED MANAGEMENT

Develop management practices for the frequency and intensity of burning to control a suite of exotic woody weeds by limiting seed production and direct mortality of juvenile and adult plants.

Background

Fire has been widely advocated as having potential as a means of addressing problems of proliferating woody plants, whether they be native or exotic species. Shrub species differ dramatically in their responses to fire. Some experience high mortality as a result of fire. They are generally classified as obligate seeders. Others are highly resilient in the face of fire and are classed as sprouters. The exotic woody species present in Australian tropical woodlands cover the full spectrum of responses to fire. For example, rubber vine and *Prosopis pallida* are fire prone while chinee apple is extremely fire resilient.

Work complete under MRC CS219, that incorporated experiments testing the effects of single and consecutive dry season fires, suggested that rubber vine could be controlled by a regime of prescribed fire. Chinee apple, on the other hand, exhibited very limited mortality even after two mid- to late dry season fires. The experiments reported here sought to build on these results to produce recommendations for the application of prescribed fire for the practical management of exotic woody weeds. The work took somewhat different approaches with rubber vine and *Prosopis pallida*, the fire-sensitive species and chinee apple, the fire-resilient species.

The experiment that was commenced under MRC CS219 and involved examining the effects of successive mid-late dry season fires on rubber vine and chinee apple, was continued. An additional experiment on the responses of chinee apple was commenced. It was designed to explore a wider range of possibilities with chinee apple, specifically, to examine its responses to a broader spectrum of times and frequencies of treatments.

Burning for the management of rubber vine and chinee apple

Methodology

The final report for Project CS219 included results of an experiment conducted at Lansdown and involving monitoring the responses of rubber vine and chinee apple to a single fire. Project 206 extended this work to examine responses of these shrubs to repeated fire and to commercial-scale fires.

At Lansdown, nine plots (each of approximately 1.3ha), were used to compare the effects of different fire regimes on the growth and survival of rubber vine and chinee apple. Between 1993 and 1999, the plots were subject to fires as shown in Figure 17.

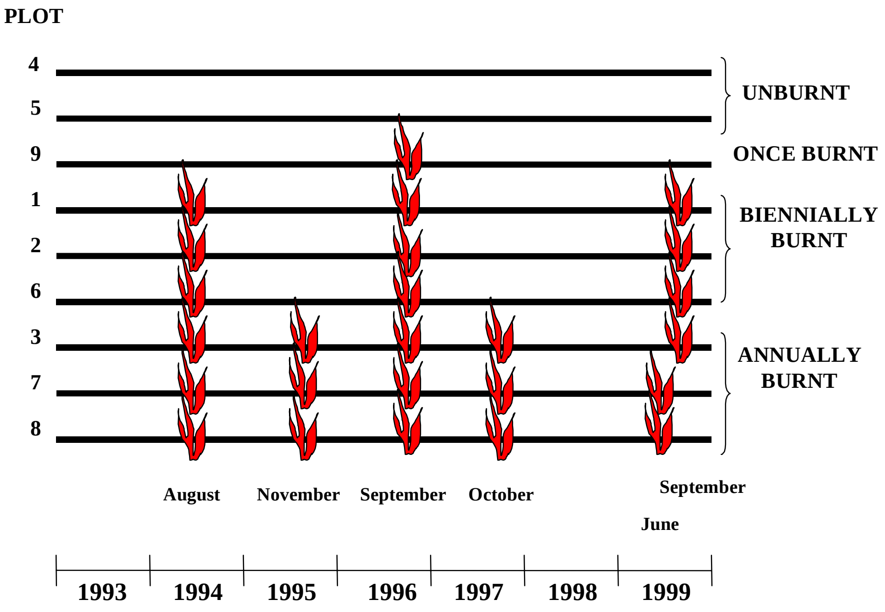


Figure 17. The times of burning on nine plots located at Lansdown Research Station.

The plan for the experiment was to compare responses of rubber vine and chinee apple to annual burning and biennial burning with that of unburnt plants. The regimes that were actually achieved differed slightly from this plan. Two plots (plots 4 and 5) remained unburnt throughout the experiment. Plot 9 was burnt once by a wildfire in September 1996. Three plots (Plots 1, 2 and 6) were burnt in 1994, 1996 and 1999. Three plots (Plots 3, 7 and 8) were burnt five times, that is every year except 1993 and 1998. No burning was done in 1998 because well-above-average rainfall meant that fuel moisture content made an intense fire impossible. Results reported here cover up until early 1999.

On each plot, 90 chinee apple and 60 rubber vine were individually tagged and numbered. These samples were made up of plants allocated to three different height classes (Table 6).

Table 6. Numbers of chinee apple and rubber vine sampled per plot at Lansdown Research Station.

Height class (cm)	Approx. no. chinee apple per plot	Approx no. rubber vine per plot
Class 1 (0-100)	30	20
Class 2 (100-199)	30	20
Class 3 (>200)	30	20

At the time the sites were established, the following parameters were measured on all tagged plants:

(i) plant height (H);

(ii) stem diameter (d); this was used to calculate stem basal area (A);
(iii) mean canopy diameter (C).

Relationships between these parameters and above-ground biomass were determined so that they could be used to document plant growth responses without destructive sampling of tagged plants. Relationships were established by measuring the same parameters on a population of untagged plants, then harvesting and weighing all above-ground material. Some of these relationships are shown in Table 7. Because plant height was much easier to measure in the field than other morphometric parameters, it was used as the index of above-ground biomass to document changes in plant size.

Table 7. Relationships between above-ground wet weight (W kg) and selected morphometric parameters.

(i) Chinee apple

$\log_{10}(W) = 0.0081(H) - 1.78$	$R^2 = 0.883$	n = 22
$W = 0.016e^{0.019(H)}$	$R^2 = 0.961$	n = 22
$W = 0.0000002(C)^{3.13}$	$R^2 = 0.977$	n = 22
$W = 0.008(d)^{2.86}$	$R^2 = 0.909$	n = 22

(ii) Rubber vine

$\log_{10}(W) = 0.0103(H) - 1.77$	$R^2 = 0.883$	n = 14
$W = 0.205(H)^{4.2}$	$R^2 = 0.933$	n = 14
$W = 0.024A^{1.5}$	$R^2 = 0.956$	n = 14

Each tagged plant was revisited every April and every October. On each occasion, plant height was recorded as well as whether it was alive or dead.

Characteristics of fires

All fires were conducted in the mid- to late dry season. The following data were collected to help characterise the fires:
(i) herbaceous biomass immediately prior to burning;
(ii) herbaceous species composition;
(iii) fuel moisture content at the time of the fire;
(iv) atmospheric temperature;
(v) atmospheric humidity;
(vi) fire speed;
(vii) fire intensity was calculated using the formula $I = H\omega r$, where I is fire line intensity, defined as heat released per metre of fire front (kWm^{-1}), H is heat yield of the fuel ($kJ\ kg^{-1}$), ω is the available fuel ($g\ m^{-1}$), and r is the rate of spread of the fire front ($m\ sec^{-1}$).

Results

For rubber vine, the nine plots differed markedly in their survivorship patterns over the course of the experiment. For chinee apple, differences were minor (Tables 8-10).

Table 8. Number of plants of height class 1 (i) chinee apple and (ii) rubbervine surviving in April of each year. Plots are listed from least to most frequently burnt.

Plot	**1994**	**1995**	**1996**	**1997**	**1998**
(i) chinee apple					
4	35	34	34	34	32
5	27	27	25	25	25
9	25	23	20	20	20
1	36	29	29	26	26
2	19	14	13	12	12
6	30	24	24	24	24
3	36	29	22	20	19
7	33	25	23	22	21
8	35	29	25	25	25
(ii) rubber vine					
4	25	24	24	24	22
5	21	20	20	20	19
9	44	38	37	1	1
1	27	0	0	0	0
2	21	1	1	1	1
6	24	0	0	0	0
3	19	0	0	0	0
7	26	0	0	0	0
8	7	1	1	1	1

Table 9. Number of plants of height class 2 (i) chinee apple and (ii) rubbervine surviving in April of each year. Plots are listed from least to most frequently burnt.

Plot	**1994**	**1995**	**1996**	**1997**	**1998**
(i) chinee apple					
4	25	25	25	25	25
5	29	29	29	29	29
9	22	22	22	22	22
1	20	20	20	18	18
2	33	32	32	32	32
6	29	29	29	29	29
3	25	23	23	23	23
7	26	25	25	24	24
8	26	25	25	25	25
(ii) rubber vine					
4	19	19	19	19	19
5	25	25	25	24	24
9	20	20	20	2	2
1	21	3	3	3	3
2	20	2	2	1	1
6	18	6	6	4	4
3	25	0	0	0	0
7	19	1	0	0	0
8	16	4	1	1	1

Table 10. Number of plants of height class 3 (i) chinee apple and (ii) rubbervine surviving in April of each year. Plots are listed from least to most frequently burnt.

Plot	**1994**	**1995**	**1996**	**1997**	**1998**
(i) chinee apple					
4	30	30	30	30	30
5	33	33	33	33	33
9	25	25	25	25	25
1	32	32	32	32	32
2	37	37	37	37	37
6	29	29	29	29	29
3	29	29	29	29	29
7	31	31	31	31	31
8	34	34	34	34	34
(ii) rubber vine					
4	15	15	15	15	15
5	12	12	12	12	12
9	18	17	17	7	6
1	12	8	8	2	2
2	19	9	9	2	2
6	17	9	9	6	6
3	16	6	4	2	1
7	16	8	3	1	1
8	7	4	1	1	1

The mortality rate of small (height class 1) unburnt chinee apple was 8% between 1994 and 1998. With a single fire, biennial, and annual burning, mortality rates for the same period were 20%, 27% and 38% respectively (Figure 18a). No chinee apple in the medium height class died in unburnt or once burnt treatments, while biennially and annual burning treatments experienced 4% and 6% mortality respectively (Figure 18b). No chinee apple in the large height classes (height class 3) died during the experiment regardless of the burning treatment (Figure 18c).

The responses of rubber vine to the various burning treatments contrasted with those for chinee apple. The mortality rate of small (height class 1) unburnt rubbervine was 11% between 1994 and 1998. With a single fire, biennial, and annual burning, mortality rates for the same period were 98%, 99% and 98% respectively (Figure 18d). Unburnt rubber vine in the medium height class (height class 2) experienced 2% mortality, while once burnt, biennially and annual burning treatments experienced 90%, 86% and 98% mortality respectively (Figure 18e). No unburnt large rubber vine died between 1994 and 1998, but once burnt, biennial and annual burning treatments experienced 67%, 79% and 92% mortality respectively (Figure 18f).

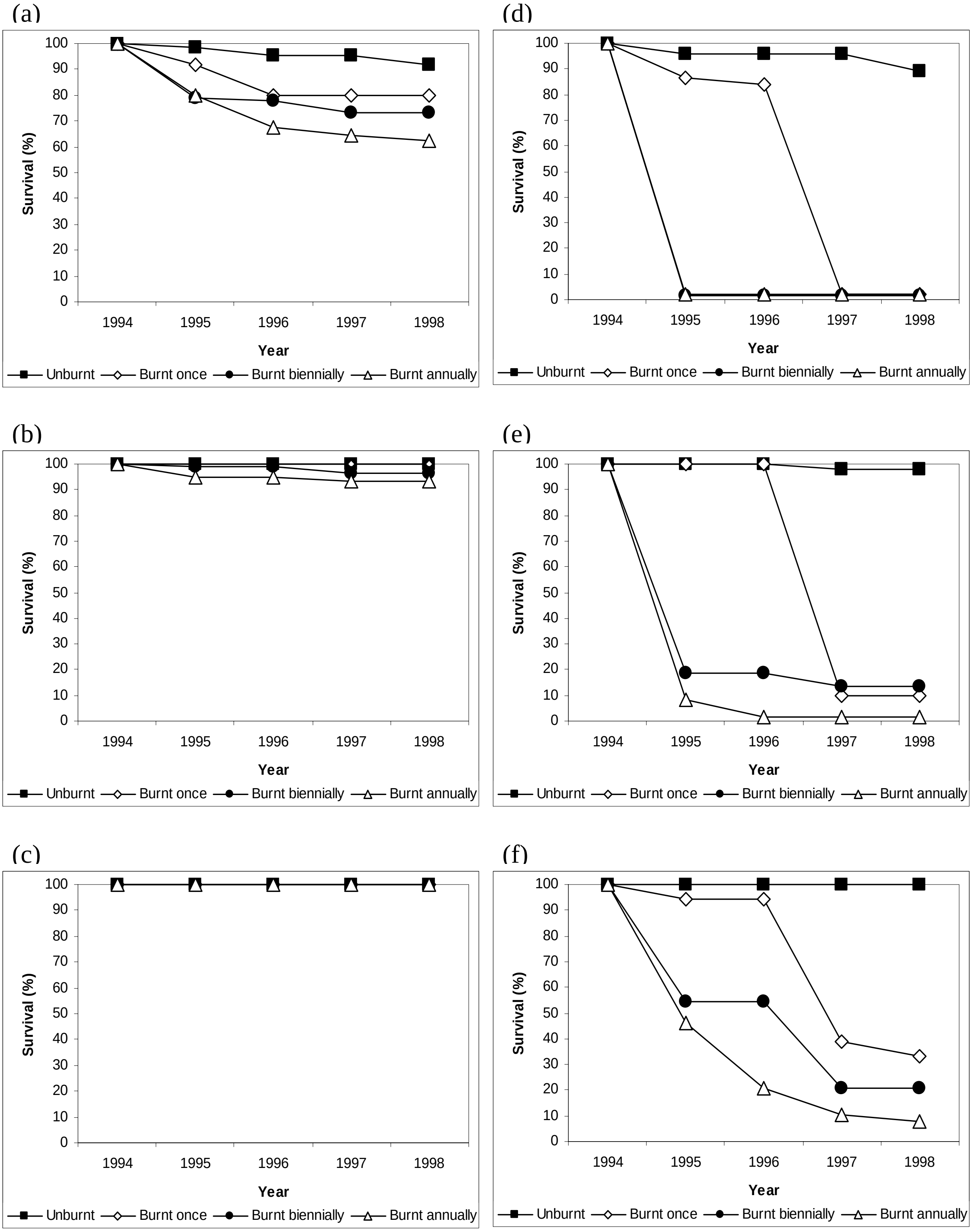


Figure 18. Survival of three height classes of chinee apple and rubber vine subject to four different burning regimes over a fire year period at Lansdown Research Station. (a) small chinee apple, (b) medium-sized chinee apple, (c) large chinee apple, (d) small rubber vine, (e) medium-sized rubber vine and (f) large rubber vine.

Characteristics of fires

The characteristics of fires at the Lansdown study site are summarised in Table 11.

Table 11. Herbaceous fuel load (kg ha^{-1}), fuel moisture content (%), atmospheric temperature (^{o}C), relative humidity (%) and fire speeds (m sec^{-1}) recorded at the time of fires conducted at Lansdown Research Station. Fire line intensity (kWm^{-1}) was calculated using data for heat yield of the fuel estimated to be 17 kJ, fuel load and fire speed.

Time of fire:	**August 1994**	**November 1995**	**September 1996**	**October 1997**
Fuel load	4160-4550	2076-3248	3507-7784	3233-8742
Fuel moisture	32-35	10-32	9-21	20-32
Atmospheric temperature	31	33	25-30	25-30
Relative humidity	35	50	20-30	60-70
Fire speed	0.3-1.0	0.2-1.24	0.02-1.29	Na
Fire line intensity	2229-18348	707-6851	49-2896	Na

Typically, the fires imposed during this experiment were high intensity grass fires, conducted when fuel moisture and relative humidity were relatively low and atmospheric temperatures were high. Grass fuel loads were also relatively high. As a consequence of this combination of conditions, fire line intensities reached quite high levels. For comparison, Luke and McArthur (1978) suggest that "the normal limits for prescribed burning are about 500 kWm^{-1}". However, a very large range of fire line intensities occurred during any one fire such that variation within a single fire was as great as variation between fires.

Conclusions

- The mortality rates of unburnt chinee apple are low. 8% of small unburnt plants died over a four year period. No mortality of medium-sized or large unburnt plants was recorded during the experiment.

- Chinee apple is highly resilient to fire and experiences very low mortality rates. Even with annual burning, mortality is restricted to plants under 1 m high, the highest rate recorded being 38% after four annual fires.

- The mortality rates recorded for unburnt small, medium and large rubber vine over four years were 11%, 2% and 0% respectively.

- Rubber vine is very susceptible to fire.

- Mortality rates of small rubbervine on burnt plots were over 95% for all burning treatments.

- Mortality rates of medium-sized rubber vine on plots subjected to a single fire were 80-90%. Four annual fires caused 98% mortality.

- Mortality rates of large burnt rubber vine were 45-67% after a single fire and 92% after four annual fires.

- On the strength of current evidence, even annual burning would do little to reduce an infestation of established chinee apple. At best, fire could be expected to make a modest contribution to chinee apple control by reducing population growth and suppressing individual plants.

- These results were achieved with fires that varied considerably in terms of fire line intensity.

- Fire can play a very significant role in the control of rubber vine. Importantly, even occasional fires would be sufficient to have a major impact upon rubber vine infestations. Two annual fires or two fires in a three-year period would drastically reduce the rubber vine population. Thereafter, periodic fire would be sufficient to prevent the species from having significant pastoral or environmental impacts. An important question that remains unanswered concerns the impacts of different fire regimes on the riparian environments that provide the primary habitats for rubber vine.

Burning for the management of prickly acacia

Previous work on *Acacia nilotica* response to fire in Africa indicates that plants greater than 50 cm tall are fire resistant under a range of fire frequencies (Pratt and Knight 1971). No quantitative data are available on fire response of seedling and early juvenile (<50 cm) though there is much evidence that fire mortality is generally high in juvenile woody species. In other work, grass fuel production has been shown to decrease markedly with increasing *Acacia nilotica* canopy cover (Carter 1994). These separate lines of evidence suggests that fire as a tool for the control of *Acacia nilotica* would be most ecologically and economically efficient early in the invasion process both in terms of seedling cohorts and in terms of adult tree density. This is so that *Acacia nilotica* seedlings do not become fire resistant and so that tree cover does not impact on grass fuel loads necessary for fires to carry.

In this report we present data on first and second year seedling fire mortality and *Acacia nilotica* canopy cover at varying tree densities. In this way we will investigate the relationship between *Acacia nilotica* size and forage production.

Methodology

The effectiveness of fire in killing seeds and first and second year seedlings was determined by conducting a controlled burning experiment in consecutive years after germination. The experiment was conducted at Wivenhoe, a property near Julia Creek. Six 70 x 40 m plots were fenced to allow grass fuel accumulation prior to each fire. In each plot three hundred seedlings from a single cohort were tagged and their height measured. Replicate plots were burnt in September 1997 and 1998, the remaining two plots being controls.

Tree canopy dimensions and density was measured at a number of sites to determine tree canopy cover at different tree densities. Methods for measurement of tree height and canopy cover are described previously (Objective 1).

Results

There is a response gradient to fire with size of plant and amount of combustible fuel (Table 12). Smallest seedlings with higher grass fuel loads have the highest mortality with fire. Conversely, larger seedlings (second year) had lower mortality with low levels of grass fuel and juvenile and adult plants suffered zero mortality in a grass fire. Both second year seedlings and juvenile plants suffered high mortality when subjected to a high intensity fire started in fallen timber.

Table 12. Mortality of *Acacia nilotica* of different sizes and with different levels of combustible fuel.

Plant size and fuel type	Plant height (cm)	Mortality attributed to fire (%)
Seedlings		
First year: 1807 kg/ha	16	99
Second year: 1439 kg/ha	24	64
Second year: fallen timber	40	90
Juvenile/adults: 1807 kg/ha	296	0
Juvenile/adults: fallen timber	310	100

Seedling growth at fire plots was faster that forecast in upland areas (see Objective 1), but slower than that calculated for bore drain seedlings (Figure 9). In optimal conditions (e.g. bore drain habitat) seedlings may become larger in a single years growth (50 cm) than they became by the second year in fire plots (31 cm).

Conclusion

Prescribed fire is very effective in reducing the density of seedling and juvenile prickly acacia.

Decapitation of chinee apple

Methodology

This experiment was designed to identify whether there are times in the annual phenological cycle of chinee apple at which plants are relatively fire-sensitive. For chinee apple plants less than two metres high, the effects of fire can be simulated by mechanical top removal because such plants respond to fire by sprouting (see MRC CS219 Final Report). By simulating the effects of fire by mechanical top removal, it was possible to conduct a tightly controlled experiment because the timing of treatments was not subject to the exigencies of the weather. With this rationale in mind, the following experiment was conducted.

The site was located within the Ross River Dam Reserve, in an extensive and relatively dense stand of chinee apple. The experiment incorporated three replicate plots and thirteen treatments. In each plot, 130 plants were individually tagged and numbered. All plants were between 100 cm and 200 cm high at the start of the experiment. They were randomly allocated to treatments, 10 plants per treatment. The treatments consisted of 12 times of decapitation. Each month from August 1997 to July 1998, thirty plants (3 plots x 10 plants) were decapitated. Decapitation consisted of manual removal of all above-ground material. Thirty plants (3 plots x 10 plants) were left intact for the duration of the experiment.

Above-ground biomass of all plants to be used in the experiment was estimated non-destructively in July 1997. This was done by relating visual estimates to the above-ground dry weight of harvested standards (Andrew, Noble and Lange 1979). A similar technique was used to estimate the initial leaf biomass (dry weight) of each shrub. Subsequent to July 1997, the total above-ground biomass and leaf biomass of each shrub was estimated at monthly intervals to document biomass recovery after decapitation. At weekly intervals after decapitation, plants were inspected for new sprouts. The presence/absence of sprouts was recorded for up to 16 weeks following decapitation. The number of fruits produced by each plant was estimated by counting fruits present at the time of fruit maturity in August 1998 and August 1999.

Photosynthetic rates of selected recently-expanded leaves of control and decapitated plants were recorded using a portable gas exchange system. These measurements were restricted to plants that were decapitated in September 1997 and January 1998 and were taken at monthly intervals for a period of one year.

Two additional treatments were used in an attempt to indirectly measure the non-structural carbohydrate reserves of decapitated plants. In addition to those plants referred to above, a further 90 plants were decapitated in September (late dry season) 1997 and 90 others in January (mid wet season) 1998. These plants were then allowed to regrow. At monthly intervals after decapitation, six plants were randomly selected without replacement and decapitated a second time, then covered by an opaque tube to prevent photosynthesis. All growth on these tubed plants was periodically harvested and the accumulative yield used as a measure of the plant’s stored reserves.

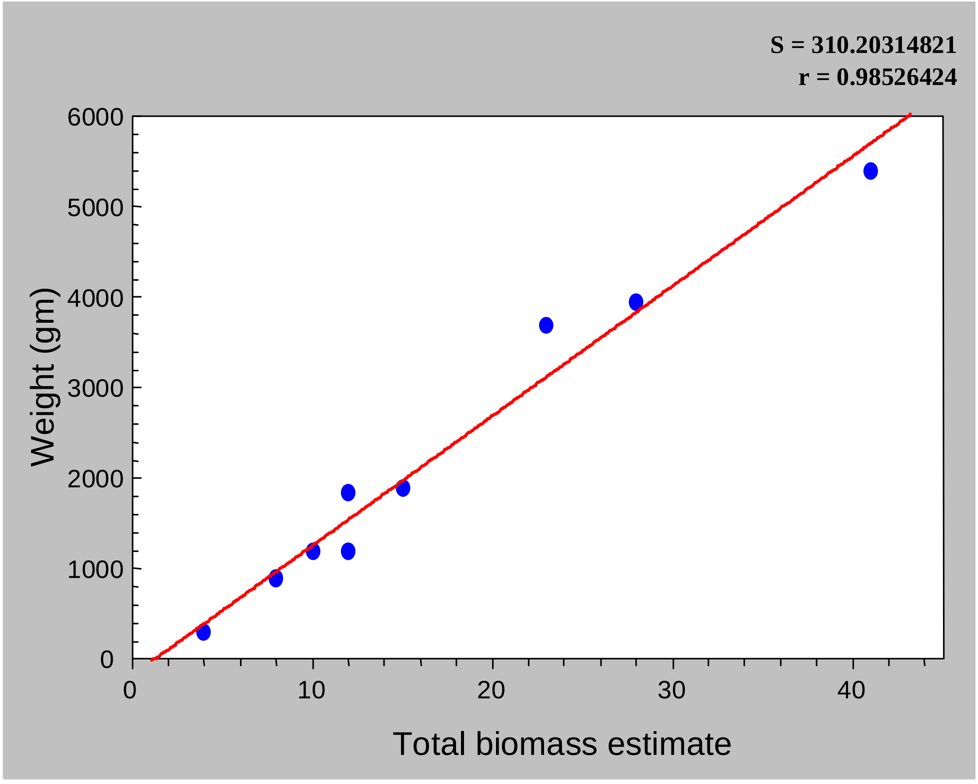
Results

The method used for non-destructively estimating the above-ground biomass of chinee apple proved sufficiently reliable for the purposes of this experiment. The equations used to relate estimated and actual above-ground dry weight were either linear (Figure 19a) or quadratic (Figure 19b) (Table 13).

Very little mortality resulted from decapitation. Only eight plants that were decapitated during the experiment failed to sprout within 12 months of decapitation. The overall mortality due to total top removal was 2.2% (n=360).

The timing of sprouting by decapitated plants varied greatly between treatments. The extremes were plants decapitated in September 1997 and February 1998. Plants decapitated in September took up to 18 weeks before sprouting commenced though some plants in this treatment sprouted within 1 week of decapitation. On the other hand, all plants that were decapitated in February 1998 had sprouted by 2 weeks after decapitation. The twelve times of decapitation fell into two groups. Plants decapitated during the wet season or early dry season (December-May) sprouted rapidly with all plants that eventually recovered doing so with 4 weeks of decapitation. Plants decapitated during the dry season (June-November) took longer, on average, to sprout. These results are summarised in Figure 20.

(a)



(b)

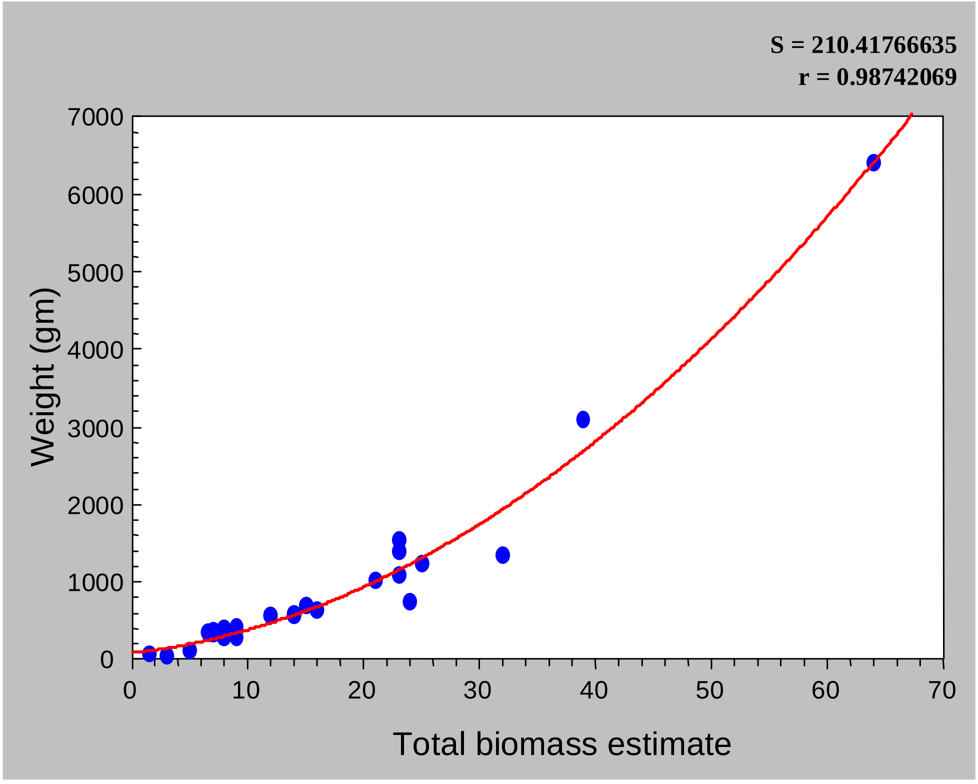


Figure 19. Examples of (a) linear and (b) quadratic relationships between actual and estimated above-ground biomass of chinee apple.

Table 13. Relationships between estimated (x) and actual (y) above-ground dry weight (grams) derived from harvested standards.

Time of Estimate	Relationship	R^2
August 1997	$y = -56649+78.05x$	0.943
September 1997	$y = 0.006+0.008x$	0.918
October 1997	$y = 0.47+0.16x+0.002x^2$	0.998
November 1997	$y = -2.09+5.36x$	0.990
December 1997	$y = 1.97+2.45x$	0.987
January 1998	$y = 3.80+5.38x+0.24x^2$	0.975
February 1998	$y = 6.48+0.24x+0.02x^2$	0.860
March 1998	$y = 34.17+18.03x+0.58x^2$	0.989
April 1998	$y = -257.87.29x$	0.956
May 1998	$y = -39.39+22.84x$	0.936
June 1998	$y = 172.27+64.89x+1.69x^2$	0.963
July 1998	$y = 12.02+97.49x$	0.848
August 1998	$y = -97.99+29.13x0.26x^2$	0.856
September 1998	$y = 87.21+17.21x+1.28x^2$	0.975
October 1998	$y = 30.68+28.51x+0.55x^2$	0.940
November 1998	$y = -165.52+59.29x$	0.879
December 1998	$y = -60.16+49.74x+0.29x^2$	0.963
January 1999	$y = -30.84+26.27x$	0.960
February 1999	$y = -61.63+58.17x$	0.955
March 1999	$y = 2.36+29.32x$	0.948
April 1999	$y = -19.20+32.64x$	0.985
May 1999	$y = 53.87+23.37x$	0.971
June 1999	$y = 95.52+38.63x+0.81x^2$	0.953
July 1999	$y = 205.48-0.40x+3.71x^2$	0.829
August 1999	$y = -52.19+47.27x+0.21x^2$	0.917

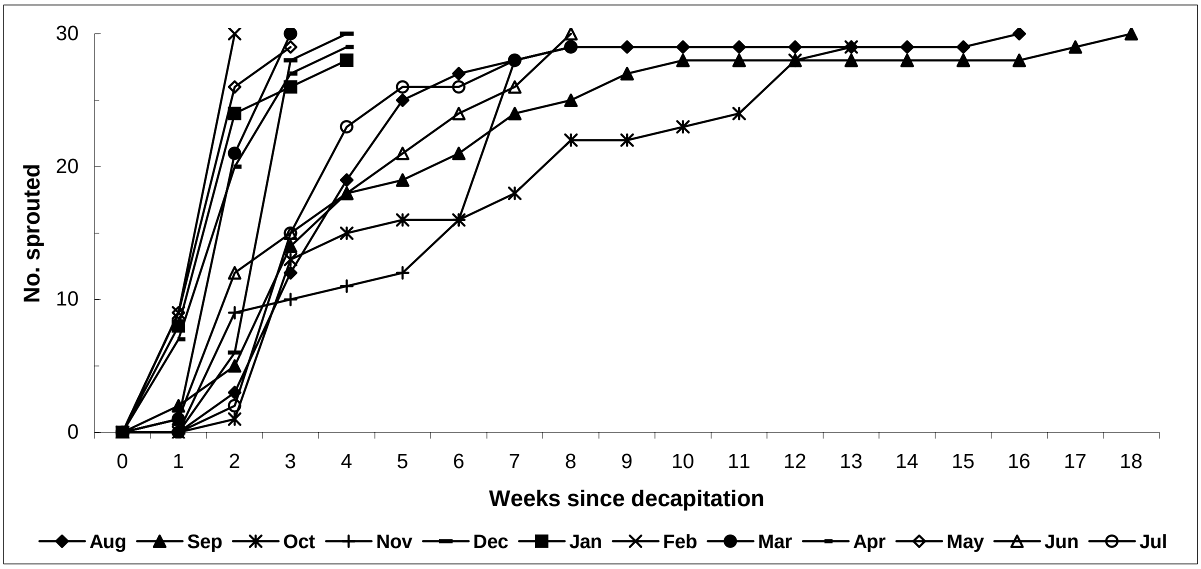


Figure 20. Number of plants sprouted at different times (weeks) after decapitation in indicated months. Treatments diverge into those decapitated in the wet season (December-May) that sprout rapidly, and those decapitated in the dry season (June-November) that sprout relatively slowly.

Temporal patterns of recovery of above-ground biomass are shown in Figure 21. Timing influenced the rate of above-ground biomass recovery. At 6 months after decapitation, the average above-ground biomass of plants decapitated in the mid-late dry season was greater than that of those decapitated in the wet or early dry season. Within 12 months of decapitation, plants decapitated between August and December had, on average, recovered their pre-treatment above-ground dry weight *(ca.* 1400g). Figure 22 illustrates further the divergence in temporal patterns of biomass recovery by plotting total above-ground dry weight as a function of time since decapitation for a limited set of treatments. Plants decapitated in August or December recovered their above-ground biomass more rapidly than those decapitated in February or April.

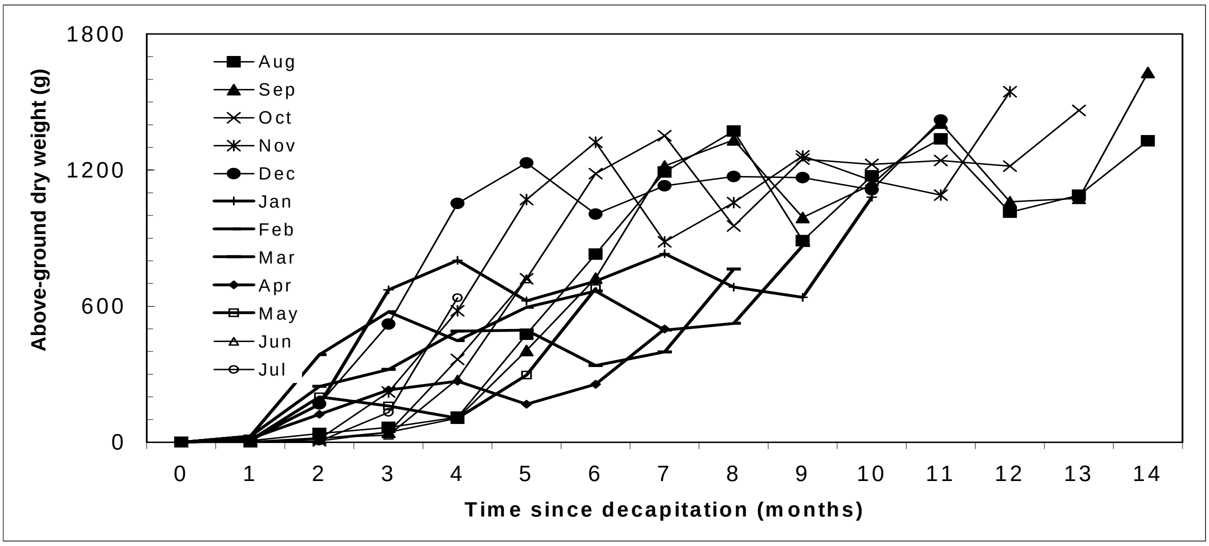


Figure 21. Above-ground dry weight at different times (months) after decapitation in indicated months.

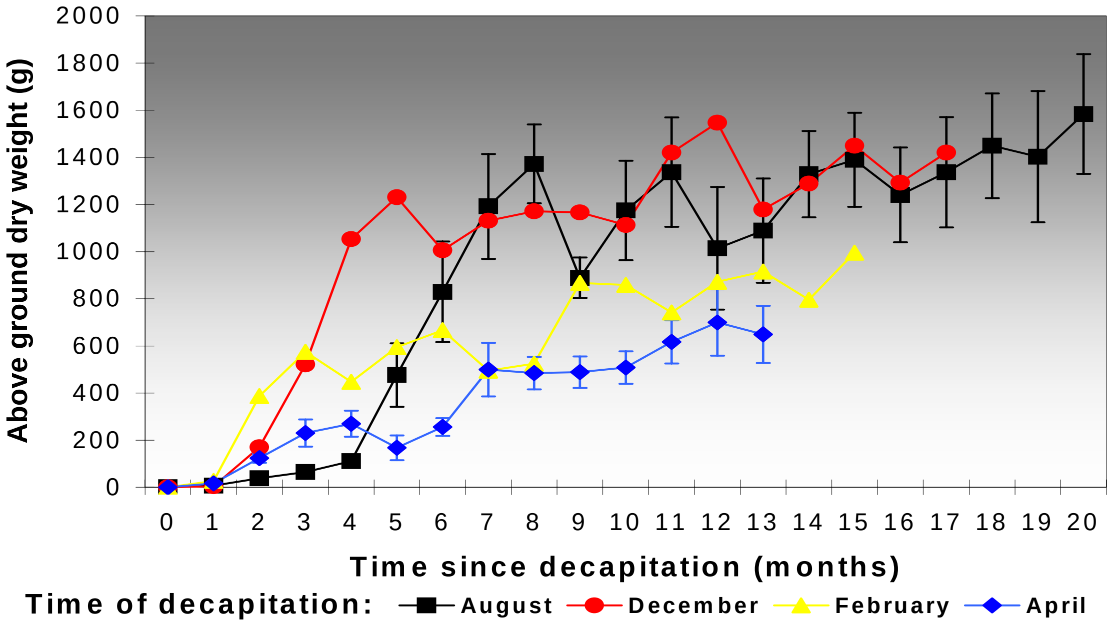


Figure 22. Recovery of above-ground portions of chinee apples decapitated in August, December, February and April.

Fruit production was affected by the timing of decapitation. In both years, fruit production was greater in plants decapitated before January 1998 (Table 14). In several instances, plants decapitated early in the experiment produced more fruit than intact (control) plants.

Table 14. Mean number of fruits produced in 1998 and 1999 by plants decapitated at different times.

Time of decapitation	No. fruits produced	
	1998	1999
August	63	18
September	91	27
October	245	23
November	198	25
December	355	187
January	8	0
February	3	6
March	0	0
April	0	0
May	0	10
June	0	17
July	0	7
Control	223	72

Decapitation dramatically altered the temporal patterns of photosynthesis. Leaves of plants decapitated in September were photosythetically active during the dry season when control plants had shed their leaves. Leaves of plants decapitated in January achieved photosynthetic rates similar to those of intact plants within 2 weeks of decapitation but maximum photosynthetic rates of regrowth were lower than control plants (Figure 23).

Results of those treatments in which plants were left to regrow in the dark show that decapitated plants had sufficient stored reserves to continue regrowth in the dark for a considerable period of time (Figure 24). Plants decapitated in September 1997 were still producing shoots in the dark in July 1999. Plants decapitated in January 1998 produced considerably less material than plants decapitated in September though the responses from the September plants varied greatly with the length of time that they had spent in the dark. Interestingly, plants that were decapitated again and tubed soon after their first decapitation produced more material than those that were allowed to regrow for a longer period (Figure 24).

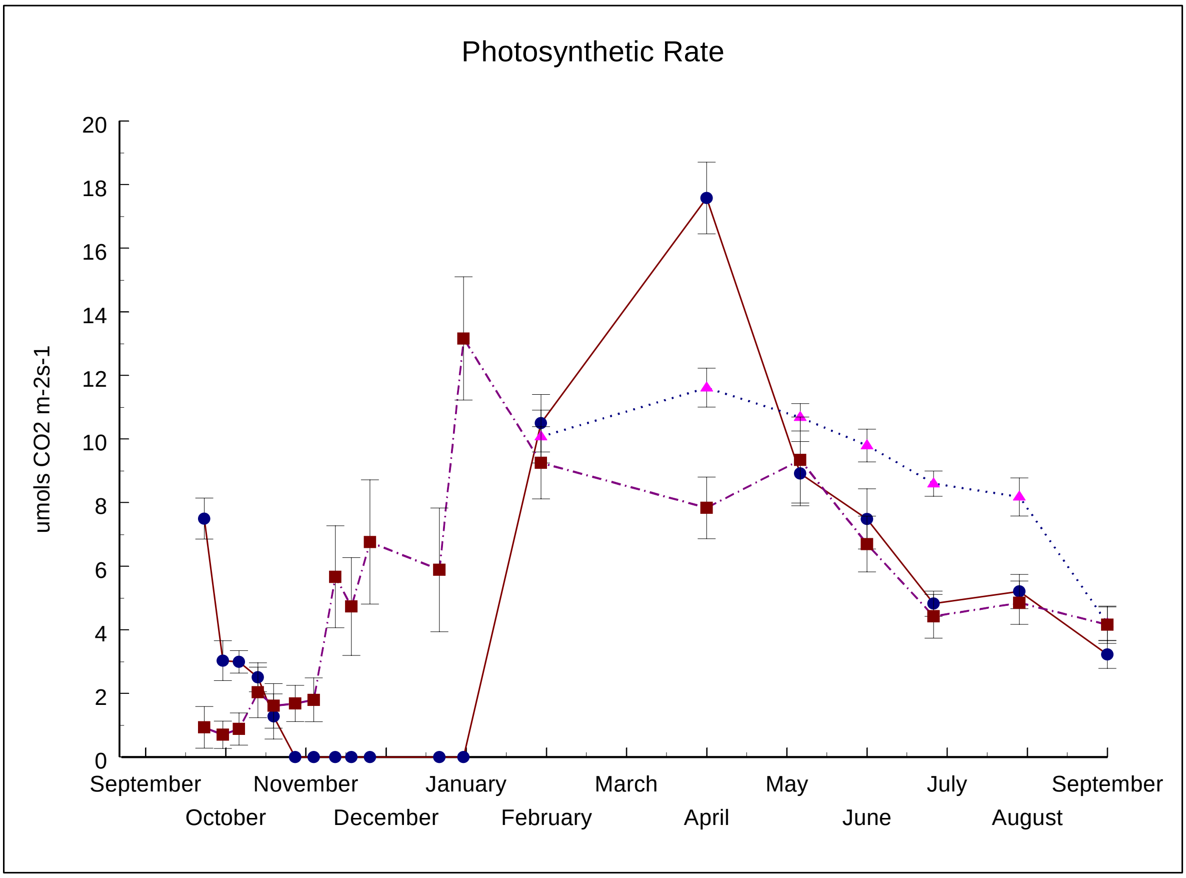
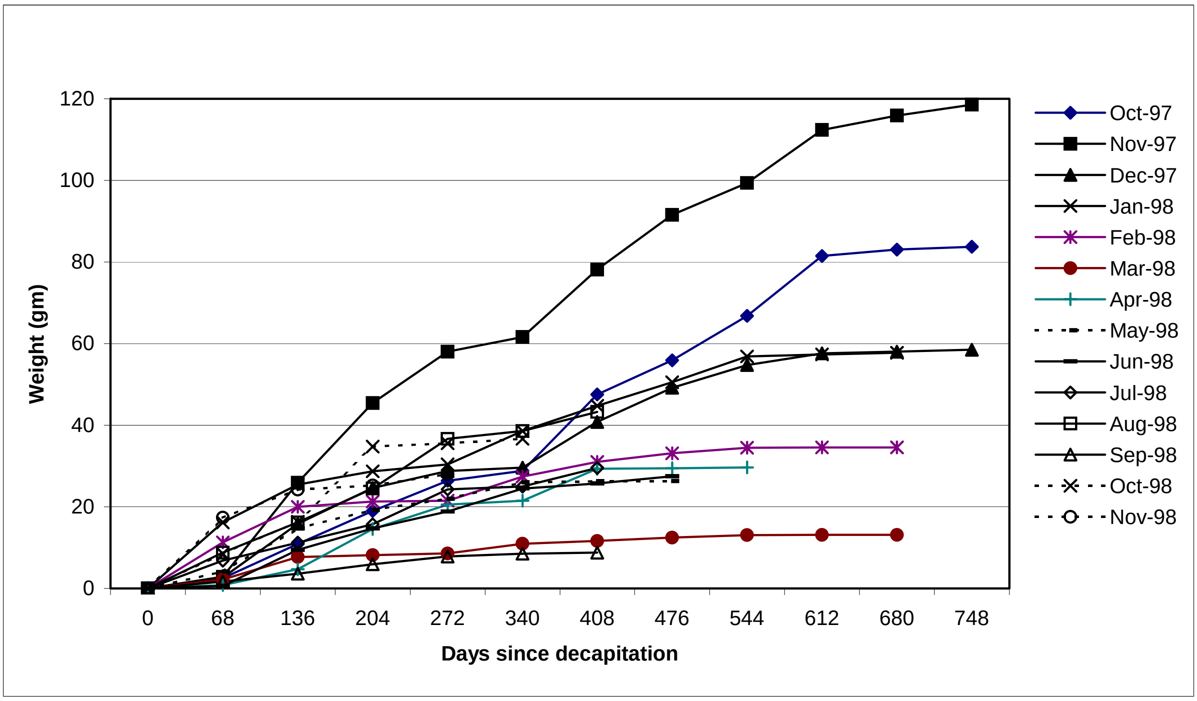


Figure 23. Photosynthetic rates of individual leaves of chinee apples. Plants were either intact (λ), or decapitated in either January (σ) or September (ν).

Conclusions

Decapitation of chinee apple has few lasting effects regardless of the time of year when decapitation occurs. This result is consistent with those of burning experiments. This experiment demonstrates the capacity of chinee apple to recover from extreme damage to the above-ground parts of the plant. It seems unlikely that a single top removal significantly reduces the non-structural carbohydrate reserves of this species. Though biomass recovery may be slower after dry season fire or top removal, after twelve months, differences between treatments are small. There is no suggestion in these results that changing the timing of a single treatment would be of any great advantage in terms of plant mortality. The only benefits that are likely to accrue from the use of fire against chinee apple are in terms of reduced seed output and, perhaps, greater access to metabolically active tissue as a means of improving the effectiveness of foliar herbicides.

(a)



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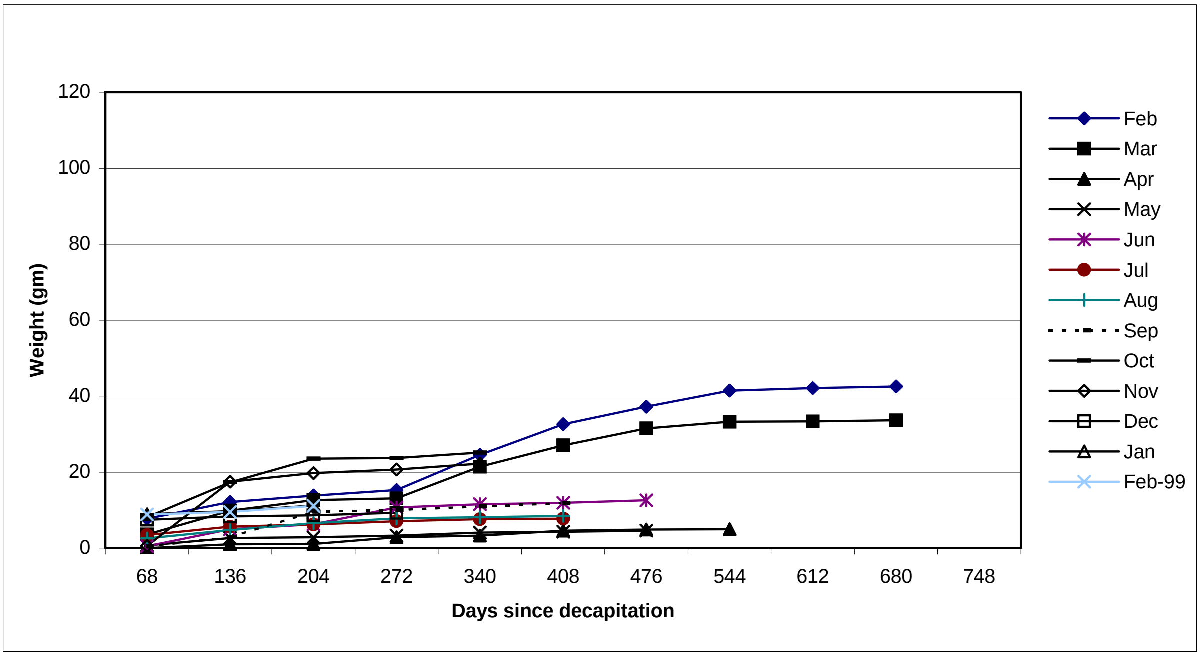


Figure 24. Accumulative biomass (above-ground weight in grams) for chinee apple decapitated in (a) September 1997 and (b) January 1998.

Susceptibility of *Prosopis pallida* (mesquite) to fire

A study was also undertaken to quantify whether *Prosopis pallida*, the most widespread of the *Prosopis* spp. in Queensland was susceptible to fire. No fire research was undertaken on *Prosopis* spp. under MRC CS 219, as the current literature suggested that *Prosopis* spp were highly resistant to fire and as a consequence burning was not considered a realistic management option. It was only following observations of some fires in northern Australia that research commenced.

Methodology

To quantify the susceptibility of *P. pallida* to burning a field study was undertaken in northern Queensland. Changes in plant density and total stem basal area were compared between burnt and unburnt plots. The experiment was conducted near Hughenden (20°51'S, 144°12'E) in north-west Queensland between October 1996 and October 1998. One month prior to burning in October 1996 a belt transect 50 m long and 10 m wide was established within each plot and the location, stem basal diameter and height of all *P. pallida* plants was recorded in each transect. At three monthly intervals thereafter until October 1998, similar measurements were recorded in all transects, with any new seedlings measured and plant mortalities noted.

Results

A single spring burn significantly decreased the density of *P. pallida*. Three months after treatment only 8% of the original density of 1760 plants per hectare remained alive in burnt plots, compared with a 100% survival rate for plants in control plots (Figure 25). Over the ensuing 18 months, a further 2.5 and 21% of the original plants died in burnt and control plots respectively. Ninety percent of control plants that died were young, less than 0.5 m in height.

All *P. pallida* plants were susceptible to fire irrespective of size. Most of those remaining alive had either not been burnt or received only minimal damage (partial leaf scalding and/or partial stem damage).

For the two years after treatments were imposed, a total of seven rainfall periods suitable for *P. pallida* germination occurred; three during the 1996/97 wet season and four during the 1997/98 wet season. Following the first significant rainfall event after burning (180 mm of rainfall over a 9 day period in January 1997), emergence results indicate that burning did significantly decrease the number of seedlings with 4000 and 2400 present in control and burnt plots respectively (Figure 25). Similarly, in the second year, significantly more seedlings were recorded in control plots. One thousand seven hundred and thirty five seedlings per ha were present in February 1998, 5 times more than that in burnt plots. By November 1998, most seedlings that had emerged over the previous two years had died, with only 60 and 16 seedlings per ha remaining alive in control and burnt plots respectively.

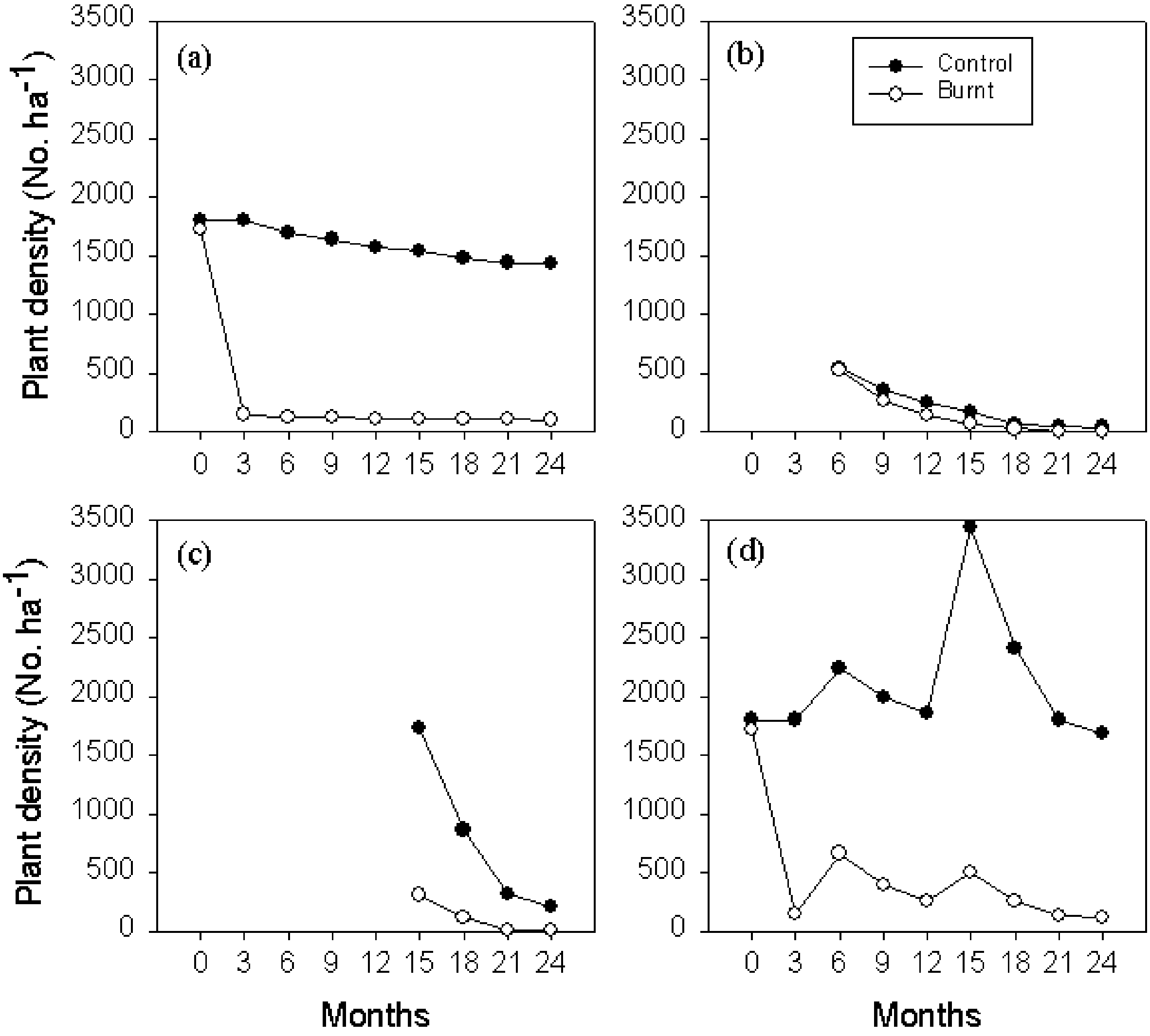


Figure 25. Changes in the density of (a) original, (b) 1996/97 seedlings, (c) 1997/98 seedlings and (d) all *P. pallida* plants over a two year period in burnt and control plots.

As for plant density, a single spring burn significantly decreased the stem basal area of *P. pallida* (Figure 26). Two years after treatment, the stem basal area of *P. pallida* in burnt plots was 0.9 m^2ha^{-1}, 12 times less than that in unburnt controls. Seedling recruitment had a minimum impact on stem basal area in both burnt and control plots, contributing less than 0.005%.

Conclusion

These results suggest that fire may be an additional control technique available to land managers for controlling the invasive *P. pallida*, provided they implement grazing practices that will enable sufficient fuel to accumulate in order to carry a fire. Plants of *P. pallida* were highly susceptible to fire, irrespective of their size. In addition, post fire recruitment of *P. pallida* in burnt plots was less than that in unburnt controls. The reason for this in the first year after burning can be most probably attributed to fire directly killing a percentage of the seed bank, most likely those seeds located on or close to the soil surface. The significant decrease in the second year on the other hand, is because of the large number of reproductive trees killed by burning, resulting in less plants contributing to the seedbank.

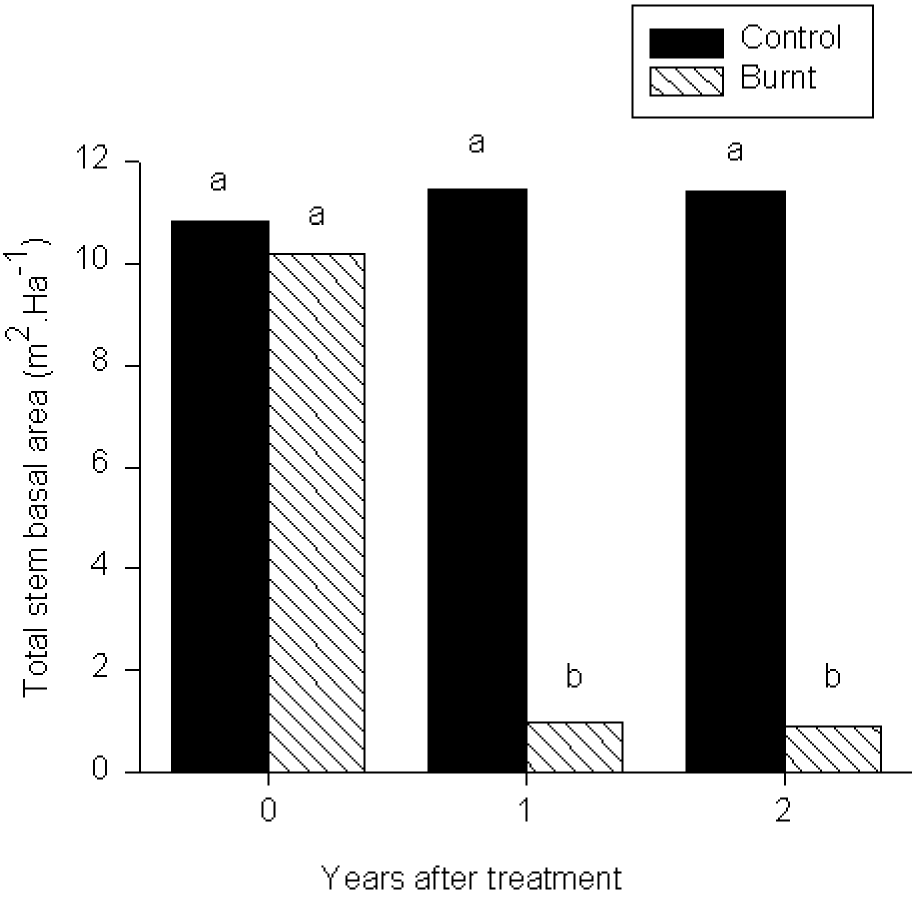


Figure 26. The Stem basal area of *P. pallida* in burnt and control plots, one month before, and 12 and 24 months after burning.

OBJECTIVE 4: DECISION SUPPORT FOR PRICKLY ACACIA CONTROL

Develop a computer based decision support package for use by research and extension officers to assist property managers to evaluate ecological and economic priorities for prickly acacia control at the paddock scale

Background

Prickly acacia is a serious weed in northern Australia from both an agricultural and conservation perspective. It is estimated that the species covers approximately 7 million hectares and costs $4.3 million per annum in lost pasture production, increased mustering costs and other costs (Brown *et al.* 1996). Prickly acacia also has environmental costs, resulting from the change from grassland to open woodland and thicket (Mackey 1998).

Much research has been undertaken on this species, in relation to control methods and ecology but, due to the scale of prickly acacia infestations and the expense of control, prickly acacia remains a serious problem. The likelihood is that there is potential for the prickly acacia problem to become much worse in the future. An understanding of prickly acacia ecology is required if effective control of this species is to be achieved.

Although we have some information on prickly acacia ecology that relates to the effectiveness of control, the practical application of those results is difficult. Data exist on seed production, seedling emergence and establishment, habitat specific juvenile and adult growth rates, phenology and mortality. Different phases of the life cycle interact with a number of environmental factors to determine population behaviour with or without management inputs. It is therefore difficult to predict outcomes of particular management decisions with multiple data sources and in specific circumstances.

In an attempt to tackle the problem of using complex data to make predictions about management options, a computer based decision support package has been developed. This package models the growth of prickly acacia and the economic outcomes using all available ecological and economic data. By inputting prickly acacia population information for particular paddocks, a producer or adviser can test the effectiveness of different control strategies on those paddocks.

Methodology

Model description

The prickly acacia model was created using DYMEX ™ (CSIRO Entomology), a generic population-modelling tool that uses climate variables and environmental parameters to drive life cycles. It does this by applying functional relationships to the variables and parameters to maintain life history properties of the average individual in each cohort on a weekly time-step.

The model for prickly acacia simulates events in a paddock through the use of two interconnected life cycles, representing bore drain and upland plant populations. The connection between the two populations is via seed dispersal by cattle or sheep. The two life cycles were necessary because in most parts of Queensland where prickly acacia occurs, paddocks contain open bore drains (artificial channels fed by artesian bores). Trees growing alongside these drains have markedly different growth rates, seedling and juvenile survival rates, fecundities, plant densities and canopy covers than trees growing in adjacent upland habitat. The only difference between the modelled upland and bore drain life cycles is the way their soil moisture environments are described: the bore drain soil moisture module includes an irrigation component.

The model identifies six discrete life stages: seedbank, seedling, juvenile, adult, flowers and seeds-in-pods. The flowers and seeds-in-pods are *endostages* i.e., lifestages that are *contained within* another lifestage – in this case within the adult life stage. This approach allows the model to deal with processes of selective abortion of flowers and pods in response to frost or drought, as well as differential development rates for pods depending upon their cohort-specific experience of environmental conditions.

The model incorporates size-dependent intraspecific competition. The surface roots of trees in the uplands frequently extend well beyond the canopy perimeters. Upland adult trees are widely spaced with respect to their canopies, though seedling and juvenile densities can be quite high. It was assumed that plants compete exclusively for space well outside their canopies during dry periods, but can tolerate higher densities during wetter periods, and at moist sites (e.g., alongside bore drains and at coastal sites).

The life cycles include mechanisms for germination, seed decay, plant desiccation, growth, maturation, density-dependent effects on growth and mortality, reproduction (including selective abortion), herbivory, interspecific competition with pasture species, and differential seed dispersal due to stocking patterns.

The model is driven by a minimum meteorological data set of weekly averages of daily maximum and minimum temperature, and weekly totals of rainfall and evaporation (Queensland Centre for Climate Applications, 1998).

The model incorporates the effects of climate change through mechanisms affecting the growth rates of plants, and their water-use efficiency. The atmospheric carbon dioxide concentration (CO_2), climate sensitivity (increase in global temperature at 2 x CO_2), and rainfall can each be adjusted to simulate future climate scenarios. The mechanisms accounting for effects of alterations in CO_2 were derived from a literature review of the effects of CO_2 singly, and in concert with temperature, nutrient and moisture availability on plant factors (photosynthesis, growth rates, water use efficiency, and competitiveness).

Where possible, driving functions and parameters were derived from published empirical relations and manipulative experiments, inferred from the plant's geographic distribution, or derived from analyses of field data. Remaining parameters were derived through *ad hoc* iterative parameter-fitting techniques.

Applications and validation

This model can extrapolate ecological data to investigate the long-term population dynamics of weed populations. It can test the effects on populations dynamics of management strategies. Economic functions can also be incorporated to look at costs and benefits of various control methods.

Several data sets can be used for validating this model. However, it is unlikely that we will be able to formally validate the model in the strict scientific sense, due to the lack of a comprehensive data set covering all aspects of the system. We can however, use comparative data generated from field monitoring and experiments. Ecological data available for comparative validation of model simulations of prickly acacia life history and ecology are numerous. Data include seed banks, seedling emergence and density, growth rates, phenology, reproduction and response to a number of control methods.

1) Applications: thresholds for the decision to control prickly acacia

The prickly acacia model was used to simulate two paddock invasion scenarios in central western Queensland. Cattle stocking rate was a constant 8 ha/head throughout both model runs. In these simulations the bore drain constituted 9.3 ha and the uplands 1239 ha, mimicking a study paddock near Hughenden. The simulation covered a 35-year period from 1957 to 1992. Suitable weather data are available for this period and many paddocks were being invaded in this region at this time. The modelled prickly acacia basal area values were used to calculate pasture production using the regression model of Burrows *et al.* (1990). While this procedure ignored inter-annual variability in pasture production, the aim of this simulation was to highlight the temporal pattern of pasture production impact due to the prickly acacia invasion rather than to quantify production *per se*. The pasture production impact function should therefore be considered an index of average impact.

Under the first scenario, the model was initialised with 2,000 seeds in the bore drain habitat. This simulated a situation where cattle carrying seed from an infected paddock to the study (target) paddock subsequently defaecate seeds near the water source. The second scenario illustrated a situation where seedlings have been deliberately introduced along bore drains. In this scenario the model was initialised with 1,000 minimum-aged adult trees in order to illustrate subsequent invasion dynamics.

2) Applications: Modelling the impact of control of bore drain prickly acacia populations

In this section we present outputs of the prickly acacia model as a preliminary test of a proposed management strategy. Ecological studies have highlighted artesian bore drain habitats as small but crucial elements of prickly acacia infestations which contribute the majority of seeds in most years (via cattle dispersal) to paddocks in western Queensland (Radford et al. 1999). A proposed management strategy is to destroy trees along these permanent waterways, thus dramatically reducing total paddock seed production (Radford et al. 1999). Modelling the efficacy of this management approach would allow us to make recommendations to beef producers about the most cost effective approach for managing prickly acacia populations below “problem” levels.

Starting conditions for DYMEX simulations were the present prickly acacia infestations recorded from a model paddock in western Queensland (Marathon South, Palm Tree paddock). In this paddock, bore drains were heavily infested with acacia (c. 540 ha^{-1}) and extended for 9.3 ha.

The open downs were lightly infested at c. 20 ha^{-1}, the total area of this habitat being 1239 ha. Bore drain and downs populations were run as separate models within DYMEX and were linked through the input of seeds from bore to downs. Cattle numbers in the paddock were constant throughout the simulation period at 400 head. Simulations were based on weather data from 1957 through to 1995.

Prickly acacia populations in this paddock were simulated under 2 scenarios, with and without seed input from bore drain populations (“with bore” and “no bore” respectively). The latter represented control of the bore populations of prickly acacia and follow up treatment of regrowth/seedlings.

3) Validation using field seed bank dynamics

The broad conclusions of the output from seed and population dynamics with and without seed input from bore drain populations were compared with results of seed bank dynamics simulations based on regressions of seed set and annual rainfall.

Starting conditions in experimental paddocks are defined in Table 15. These were all paddocks in which *Acacia niloitca* is well established, with trees found throughout each paddock, though at sparse density across most of the paddock area.

Table 15. Location, tree density and habitat area in experimental paddocks.

	Marathon	Proa	Wivenhoe	Keswick
Latitude-	20^{0}53.20’S	20^{0}53.24’S	20^{0}65.00’S	20^{0}22.10’S
Longitude	143^{0}41.08’E	142^{0}09.24’E	142^{0} 21.07’E	141^{0}34.29’E
Density (Upl)	20 trees.ha^{-1}	16 trees.ha^{-1}	43 trees.ha^{-1}	31 trees.ha^{-1}
Density (BD)	2665 trees.ha^{-1}	516 trees.ha^{-1}	207 trees.ha^{-1}	598 trees.ha^{-1}
Area (Upl)	1239 ha	1593 ha	865 ha	1728 ha
Area (BD)	9.3 ha	0.1 ha	0.6 ha	3 ha

Upl, upland habitat; BD, bore drain habitat.

Data collected in these paddocks were as follows. First, seed production was measured in both bore drains and uplands. Second, we measured seed dispersal across upland habitats to establish the magnitude of seed banks. Third, seed banks were modelled with and without seed input from bore drain trees over two 10 year periods, to simulate the situation where bore drain trees were controlled compared with when they were not controlled. Regressions between rainfall and seed production per tree allowed the calculation of total seed production for the paddocks in both bore drain and upland populations.

Seed banks were modeled with and without the contribution of seeds from bore drain trees, simulating a situation where prickly acacia was controlled along bore drains using currently available technology (both chemical and/or mechanical methods, March 1995). Model paddocks were based on experimental paddocks and were given the same tree densities and total area (Radford et al. 1999). Seed production per tree was generated from regression relationships with rainfall and the percentage of seed dispersed into upland habitats based on the proportion of total seed produced that was found in dung at each site. Total viable seed for each paddock was calculated using the percentage seed that was potentially viable from bore drain and upland trees. It was assumed that seeds transported into the paddock were evenly distributed. Cow dung pat counts were constant for each paddock, based on cattle numbers being the same throughout the study.

Two modelling runs were completed. Both were based on a decade of rainfall data from Julia Creek Post Office. Rainfall from the 1987/88 wet season through to the 1996/97 wet season was chosen to represent a “typical” rainfall scenario (both wet years (>500 mm) and dry years (<500 mm) occurred in this period). Treated and untreated paddocks were compared on the basis of annual seed numbers per cow pat and the accumulated number of viable cow pats (those with more than one germinable seed) per hectare. These numbers were calculated on the basis of constant annual seed decay rates derived from a seed burial experiment (Brown *et al*. 1996) and defined by the following exponential equation ($y = 552.36e^{-1.5819x}$, $R^2 = 0.97$).

The second modelling run was generated using rainfall data from a series of very dry years from 1981 to 1990 (Julia Creek P.O.). Eight out of 10 years in this decade had rainfall below 500 mm. Marathon and Wivenhoe only were modelled for this run, to represent contrasting landscape patterns; the former with an extensive bore drain area, the latter with relatively small bore drain area.

Results

Ecological and economic thresholds for management

Scenario 1 – Accidental Introduction by Stock

The modelled pattern of paddock invasion starting with 2,000 seeds deposited in the bore drain habitat, took the following course. Initial seed deposition resulted in a flush of seedlings with a high survival rate due to their moist habitat. After about one year, these developed into juvenile plants that matured at approximately 6 years of age. During this period there was no additional seed production. After the bore drain juveniles matured, seeds were produced and dispersed to both bore and upland habitats, resulting in germination following rain events. Pasture production rapidly deteriorated along the bore drain as canopy cover and basal area increased. In the uplands (that are responsible for the majority of pasture production in the paddock), there is less than a 10% reduction in pasture production 35 years after the initial recruitment event and 29 years after the first upland recruitment episode.

Scenario 2 – Deliberate Planting

The simulated pattern of paddock invasion starting with 107 trees ha^{-1} in the bore drain habitat, occurred as follows. The bore drain trees rapidly built up the seedbank, leading to episodic recruitment in the uplands. The rates of impact on the pasture production index were minimal until plants started maturing. When adult plants start appearing in the uplands, the basal area rapidly increases, corresponding with a rapid drop in the pasture production index.

Ecological impacts of strategic management strategies

We have chosen total seed production (Figure 27), juvenile and adult tree density and proportion of canopy cover (Figure 28) to summarise the results of acacia population simulations with and without seed input from bore drain trees.

Seed production figures in the two simulations progress through three stages over the course of the simulations (Figure 27a). Initially (up to year 13) seed production was lower or approximately the same for the "no bore" treatment compared with the "with bore" treatment (Figure 27a), a situation which approximates present conditions depending on annual rainfall (Radford *et al.* 1999). From year 13 to 19 a massive increase in seed production occurs in "no bore" populations, while a more gradual increase occurs in "with bore" populations from year 18 to 23 (Figure 27b). Finally, seed production fluctuates at about the same level in both simulations from year 22 to 39 (Figure 27b).

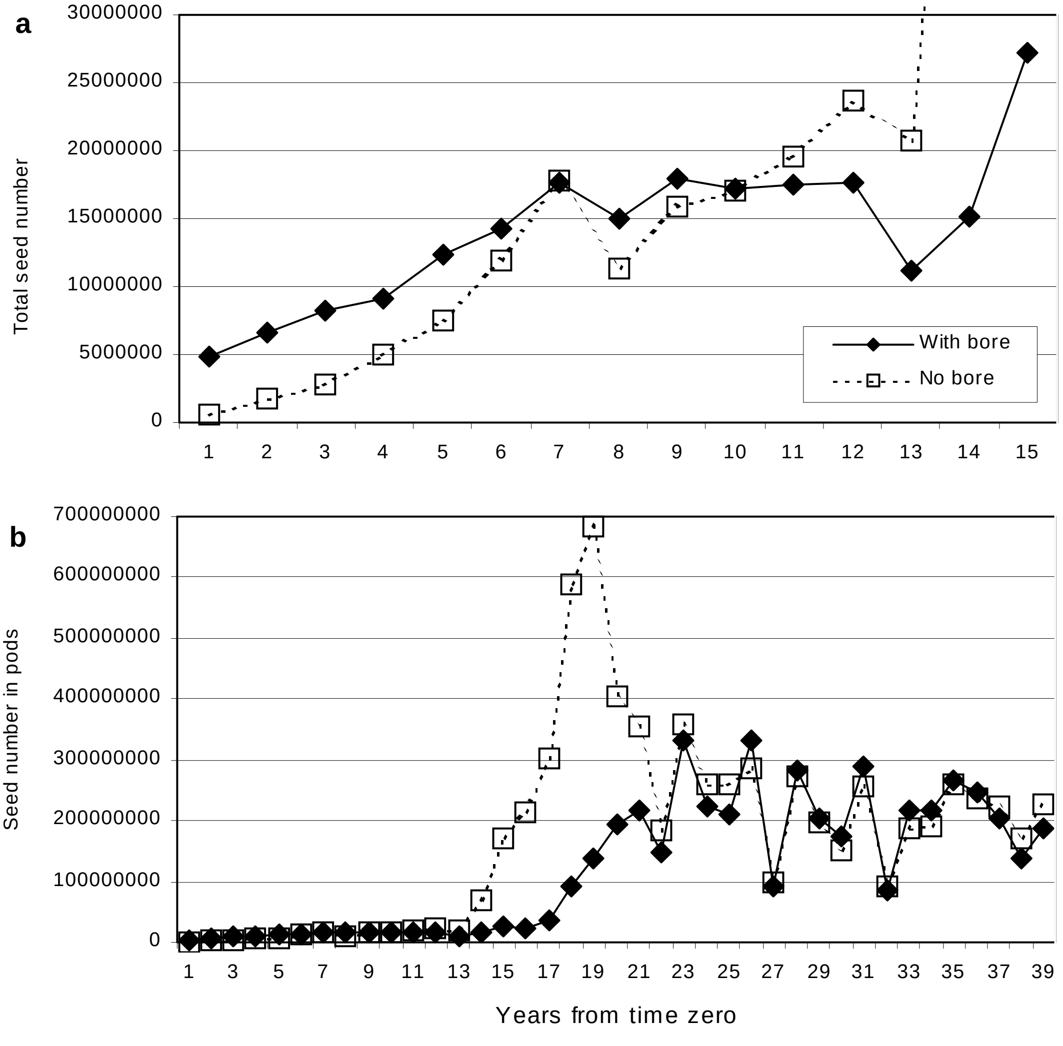


Figure 27. Seed production for Prickly acacia populations with and without bore drain tree input. **a)** Seed production in the first 15 years of the simulation and **b)** seed production over the full 39 years of the simulation. Solid lines and diamonds (___♦) indicate seed production with bore drain input, and dashed lines and open squares (-----□) indicate seed production without bore drain input. Meteorological data from 1957 to 1995 were used.

Juvenile density increased rapidly after year 7 in both "with bore" and "no bore" simulations, though much higher density is attained in "with bore" paddocks and juvenile populations are more persistent (Figure 28a). In contrast, adult tree populations initially increase more rapidly in the "no bore" simulation (Figure 28b). "With bore" populations subsequently increase to approximately double the density attained in the "no bore" simulation. From these peaks both populations decrease gradually over the course of the simulation, "no bore" populations reaching an apparently stable level of about 150 trees ha^{-1}. Figure 28c shows more rapid growth and greater total canopy cover for "no bore" populations in the initial population build-up from year 11 to 22. Total canopy cover then stabilises at its assumed sustainable upper limit (0.3 to 0.4 acacia ground cover).

Validation: seed bank impacts of controlling bore drain trees

Seed numbers per dung pat were highly variable between years and sites based on validation data (Table 16). Larger numbers of seeds were present in years with higher rainfall. Major differences were occurred between populations with and without bore drain seed input, particularly after dry years (<500 mm annual rainfall), but differences were reduced after wet years due to compensatory seed production in upland habitats. The least variable numbers of seed per dung pat was seen at Marathon, probably because there were more dung pats in the paddock. The largest effect of removing bore drain seed input was seen at Marathon because of the very large and productive bore population compared with other experimental paddocks.

Table 16. Yearly numbers of germinable seeds per dung pat in modelled acacia populations.

Year	Rainfall	Mar BD	Mar Up	Pro BD	Pro Up	Wiv BD	Wiv Up	Kes BD	Kes Up
1987/88	381	11	0.006	4	0.9	5	0.6	2	0.2
1988/89	463	17	0.03	11	5	10	4	3	0.3
1989/90	589	34	0.6	104	93	74	62	8	3
1990/91	782	134	46	7190	7161	4812	4781	219	206
1991/92	535	25	0.2	36	28	27	18	5	0.8
1992/93	435	15	0.02	8	3	7	2	2	0.1
1993/94	622	40	1	208	196	144	131	11	6
1994/95	273	7	0.0005	2	0.1	2	0.1	1	0.002
1995/96	322	8	0.002	3	0.2	3	0.2	1	0.007
1996/97	685	59	5	825	807	558	539	31	23

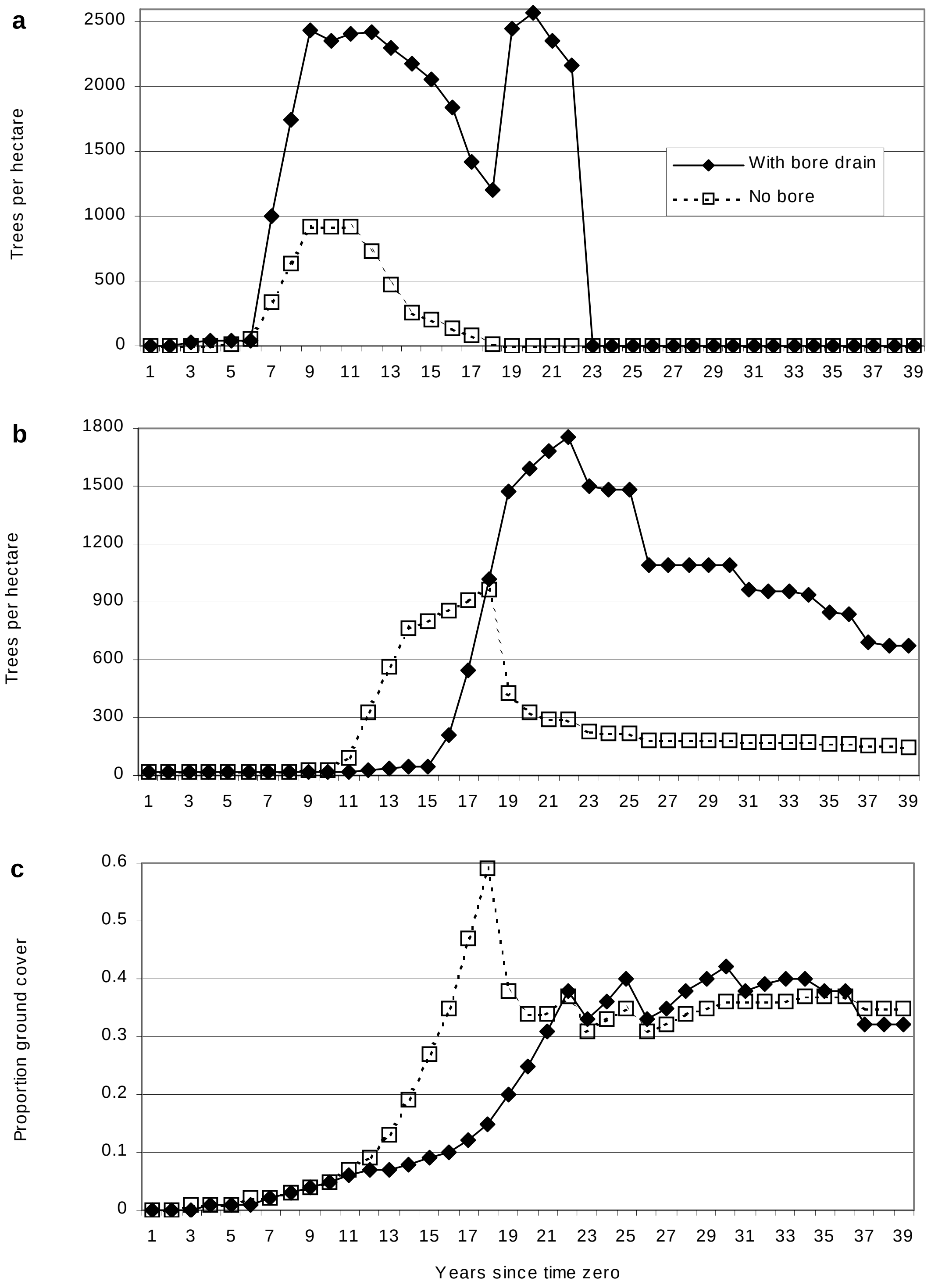


Figure 28. a) Juvenile density (ha^{-1}) , **b)** adult tree density (ha^{-1}) and **c)** proportion canopy cover through a 39 year population simulation. Solid lines and diamonds (___♦) indicate seed production with bore drain input, and dashed lines and open squares (-----□) indicate seed production without bore drain input. Weather data used was derived from 1957 to 1995 meteorological data.

In modeled populations there was a reduction in the number of dung pats with germinable seeds when bore drain population input was removed (Table 17). Differences, however, were much less than the differences in total seed production with and without bore drains. The largest reduction was seen at Marathon, where 65% fewer viable dung pats were produced. Keswick showed a 40% reduction in viable dung pats with the removal of bore trees. Proa and Wivenhoe propagule outputs were reduced by less than 20%. The number of dung pats with germinable seeds was more variable in paddocks with upland seed input only, with numbers being reduced by more than half after dry years, while paddocks with bore drain trees intact had much more stable numbers. Despite this, numbers of viable dung pats did not generally reduce dramatically due to the accumulated seed bank and the delayed loss of seed viability with respect to intervals between dry years. This is because seeds took three years to reduce to 1% viability while during the simulation period only two successive dry years were seen. Due to the seed clumping effect (through deposition in relatively few dung pats) the number of dung pats with germinable seeds was broadly proportional to the number of cattle in experimental paddocks. Proa and Wivenhoe had the lowest numbers of ‘viable’ dung pats and the lowest number of cattle (100 and 120 cattle). Keswick had intermediate numbers of ‘viable’ dung pats (with 230 cattle) and Marathon had the highest viable dung pat numbers (with 450 cattle).

Table 17. Accumulated number of acacia propagules (dung pats with more than one germinable seed) in modelled experimental populations.

Year	Rain fall	Mar BD	Mar Up	Pro BD	Pro Up	Wiv BD	Wiv Up	Kes BD	Kes Up
Start Up (%)		353 #	1 (0.2%)	129 #	87 (67%)	74 #	48 (65%)	141 #	42 (30%)
1987/88	381	385	251	143	134	90	76	211	142
1988/89	463	427	163	154	131	102	82	196	67
1989/90	589	506	185	133	112	95	83	163	88
1990/91	782	464	210	124	92	103	81	163	143
1991/92	535	467	166	115	109	107	102	199	164
1992/93	435	524	155	141	140	133	132	226	118
1993/94	622	564	274	157	120	144	142	246	152
1994/95	273	577	140	134	108	147	90	241	143
1995/96	322	487	23	152	87	142	77	208	54
1996/97	685	500	140	149	111	140	102	160	83
Mean		490	171	140	114	120	97	201	115
Percent			35%		82%		81%		57%

#Modelling simulations for each experimental site used the number of germinable propagules calculated from both bore drain and upland acacia populations.

In order to test the effect of a series of dry seasons on the number of viable dung pats, Marathon and Wivenhoe populations (with the highest and lowest number of cattle and contrasting bore drain areas) were modelled in a decade of low rainfall (1988 to 1997). Viable dung pat numbers in these dry years were compared with the previous climatic scenario (see Table 16 and 17, Figure 29). In Marathon paddock (Figure 29a) there is a greater reduction of dung pats with germinable seed in both the average and the dry climate scenario, however, only in the dry scenario is there more than one year in which propagule numbers are reduced to zero. In contrast Wivenhoe propagule populations (Figure 29b) show more difference between climatic scenarios than between populations with and without bore drain seed input. Viable dung pats were reduced below 40 ha^{-1} for three years in the dry scenario, but later recovered.

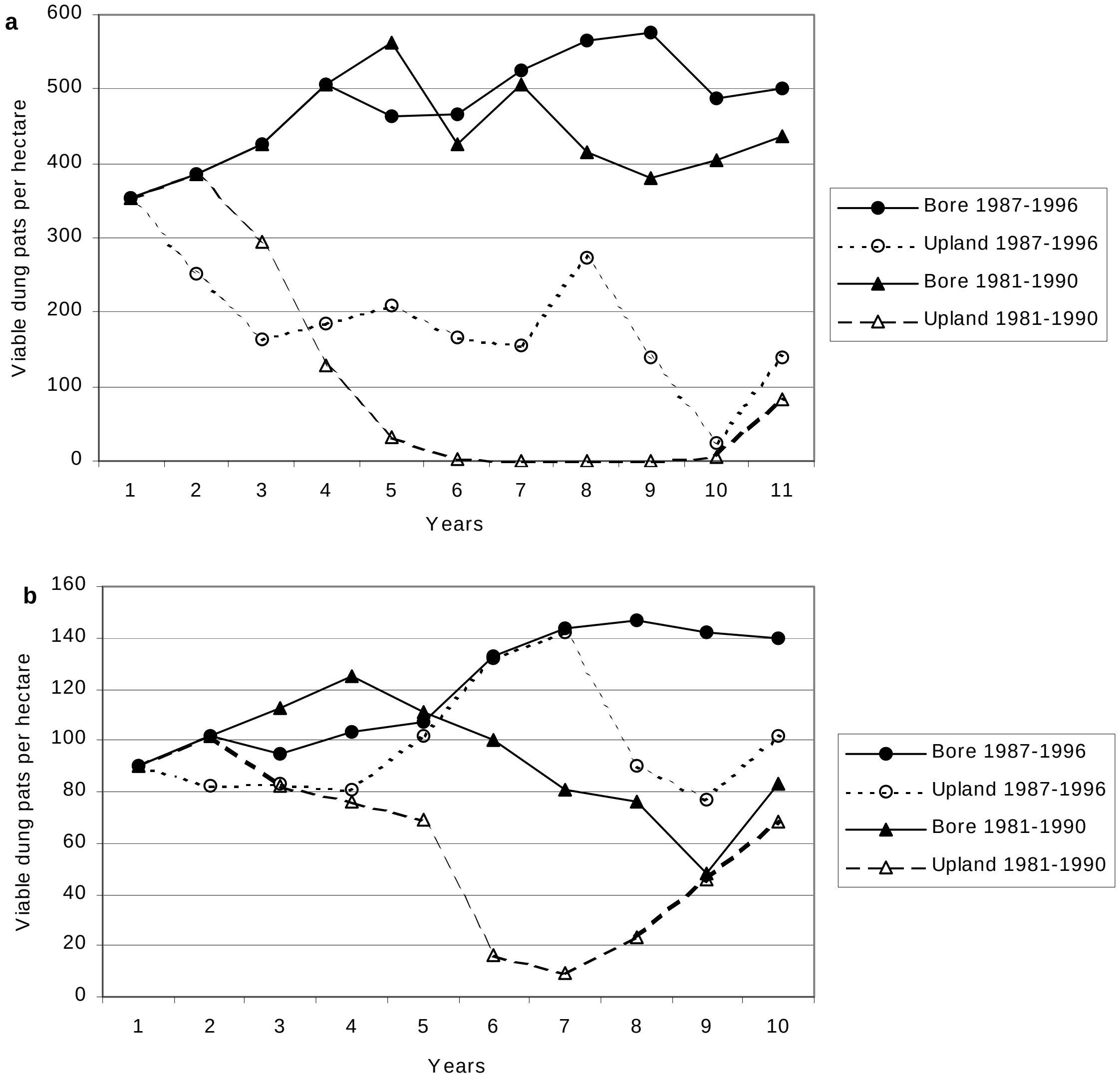


Figure 29. Numbers of viable dung pats per hectare in modelled populations of a) Marathon and b) Wivenhoe paddocks over a series of 10 dry (<500 mm annual rainfall) years.

Discussion

Ecological and Economic priorities for Prickly acacia control

For prickly acacia control to be ecologically and economically efficient, it must be undertaken early in the invasion process. Model simulations suggest that control of trees needs to be initiated at least 10 years prior to significant impacts being measurable if initial populations are small and localized, and 20 years where Prickly acacia was widely planted. Because costs of control increase linearly with the area infested, control at this early stage, when populations are small, is much cheaper (March 1995, Miller 1996). Control early in the invasion process is also much more ecologically sound because extensive seed banks have not yet been established in infested paddocks. Established seed banks make the control of prickly acacia much more difficult because seed banks may allow re-establishment of populations years after control has been undertaken.

The results of modeling paddock populations with and without bore drain trees re-enforce the importance of early control because they show there is no cheap option for controlling prickly acacia populations once established. Both the DYMEX prickly acacia model and simple seed bank modeling indicate that controlling bore drains alone will not achieve sufficient reduction in seed numbers to stop future establishment. Even though trees on bore drains individually produce seeds far in excess of upland trees (Radford et al. 1999), enough seeds were produced by upland trees across the whole paddock to allow for recruitment under the right conditions. This means that the relatively cheap option of controlling trees only in the bore drains, which cover less than 1% of the area of most paddocks (<10 ha), is not likely to be an effective control option.

It is important for producers to recognise the difference between early and late prickly acacia invasion in order to implement appropriate actions for control. Where there are only a few trees or juveniles in a paddock it is obviously early in the invasion and would be an appropriate time to implement control. Even where dense populations occur, if they occupy only a small part of the paddock (e.g. riparian or bore drain situations), control could still be highly effective. Once trees are established throughout much of the paddock, even if there are only a few trees per hectare, upland seed production becomes sufficient on its own for future recruitment. This means that control must occur across the whole paddock for it to be effective.

To compare the economics of early and late control, it is much cheaper to apply control early. It may cost only $10 to control a few trees in one paddock. Even with a dense stand of prickly acacia of one hectare, control by mechanical means could be achieved for $250 (not including any follow up treatment, March 1995). Control of bore drain populations (as described in Table 17) would mostly cost between $280 and $1,500 using Diuron, a chemical treatment applied to drains (up to $4,400 where bore drains cover large areas) (March 1995). By comparison, late invasion control of prickly acacia in the entire paddock would cost between $2,200 and $14,000 for the initial treatment (March 1995). As indicated above, established seed banks in paddocks where prickly acacia is well established would mean that follow up treatments would have to be re-applied several times for control to be complete.

Model Validation

In broad terms, the results from the DYMEX prickly acacia model and the simple model agree. The number of dung pats with germinable seeds calculated using simple modeling (between 100 dung pats and 500 dung pats/ha) are about equal with the yearly tree density increase during the population expansion period between years 11 and 22 in the DYMEX simulation (Figure 28). As previously observed, maximum tree density is determined by dung pat density due to clumping of seeds and self-thinning (Kriticos *et al.* in press). If we assume high rates of juvenile plant survival during these expansion periods, then increases in tree density on this scale are realistic. As indicated by simple modeling of populations at Marathon (Figure 29a), such recruitment would only be possible during an average to wet decade in some paddocks.

A major short-coming of our validation is that it only considers one aspect of population dynamics; the seed bank. Although there are useful data on seed production, seedling establishment (over 3 years), and growth and survival of juveniles and adults over short periods (<3 years), there are few that cover long term population dynamics or the specific conditions allowing recruitment. This is because we have only short-term data and we have not been able to study directly a recruitment event. The data currently available indicate that seedling survival to one year is mainly less than 5% in Prickly acacia populations (Radford et al. unpublished). At this level of establishment population increases observed in the DYMEX model may not have been possible.

Future studies into prickly acacia ecology must focus on the conditions required for recruitment to occur and therefore for invasion to take place. Such data should be based on either long term data sets or intensive experimental studies investigating a range of conditions in parallel. Such data are needed to allow appropriate validation of the prickly acacia DYMEX model so that we can be confident in the results of producer management based simulations.

Decision Support Package

The prickly acacia model is in the final stages of preparation before release to stakeholders. It can be used to simulate various management scenarios to give producers decision support for controlling their infestations. The availability of this model within extension agencies now relies on the work being written up by the PhD student involved in the project.

OBJECTIVE 5: INTEGRATION OF ECOLOGICAL AND ECONOMIC DATA

Integrate ecological data into economic analysis for decision support of rubber vine management at the paddock scale.

Background

There is a variety of valid techniques available for the management of rubber vine. These include application of herbicides, mechanical techniques, two biological control agents (the *Maravalia cryptostegiae* and the moth *Euclasta gigantalis*), and prescribed burning. While each is valid in the sense that it can be effective, they are not equally applicable under all circumstances. Perhaps most importantly, the cost of using a particular technique or combination of techniques will govern whether it is most appropriate.

Fire has been shown to be particularly valuable as a tool for the control of rubber vine. It can be applied on a very large scale without the need to locate and treat individual plants. In fact, fire and biological control are the only tools for rubber vine control that are genuinely large scale. However, the use of prescribed fire does not come without costs. Most obvious are the costs of constructing and maintaining fire breaks and the opportunity cost of grazing forgone in order to accumulate adequate grass fuel before burning and to allow sufficient time for perennial grasses to recover after burning.

Economic assessment

An economic assessment of the use of fire for rubber vine management was developed for cattle grazing country in the Charters Towers area. It was based on the following assumptions:

- Actual climatic data for Charters Towers for the period 1955-1995.
- Prescribed fires conducted every eight years.
- A paddock of relatively high fertility for the district, and occupying 15% of the property.
- Either set stocking at 3.6 ha/animal equivalent or variable stocking utilising 40% of available forage.
- Live weight gain calculated using GRASSMAN (Scanlan and McKeon 1990).
- Paddock rested 12 months before burning to allow fuel build up and two months after burning to allow pasture regeneration.
- Comparisons made in terms of the value of gross beef production.
- Variable costs of burning \$0.60/ha and covering fire break maintenance, labour and vehicle variable costs.
- Hypothetical rubber vine density just starting to impact on production in the early years of the simulation with density and canopy cover doubling in the first 10 years, rising to 72% cover after 40 years.
- 8% discount rate.

In line with the conclusions of MRC Project CS219, these assumptions yielded the conclusion that grass production benefits and resultant beef production responses were not sufficient to justify burning to contain rubber vine, given the climatic conditions between 1955 and 1995 and a discount rate of 8%. Sensitivity analysis of an economic assessment of the use of fire for the management of rubber vine, shows that the outcomes are sensitive to assumptions regarding destocking and discount rate.

Table 18. Net present value and benefit:cost ratio for rubber vine management using fire.

Discount Rate	Stocking regime	Net Present Value $/ha	Benefit:cost Ratio
8%	Variable	-3.76	0.96
	Set	-21.77	0.63
7%	Variable	6.26	1.06
	Set	-18.12	0.7
4%	Variable	56.97	1.48
	Set	1.87	1.03

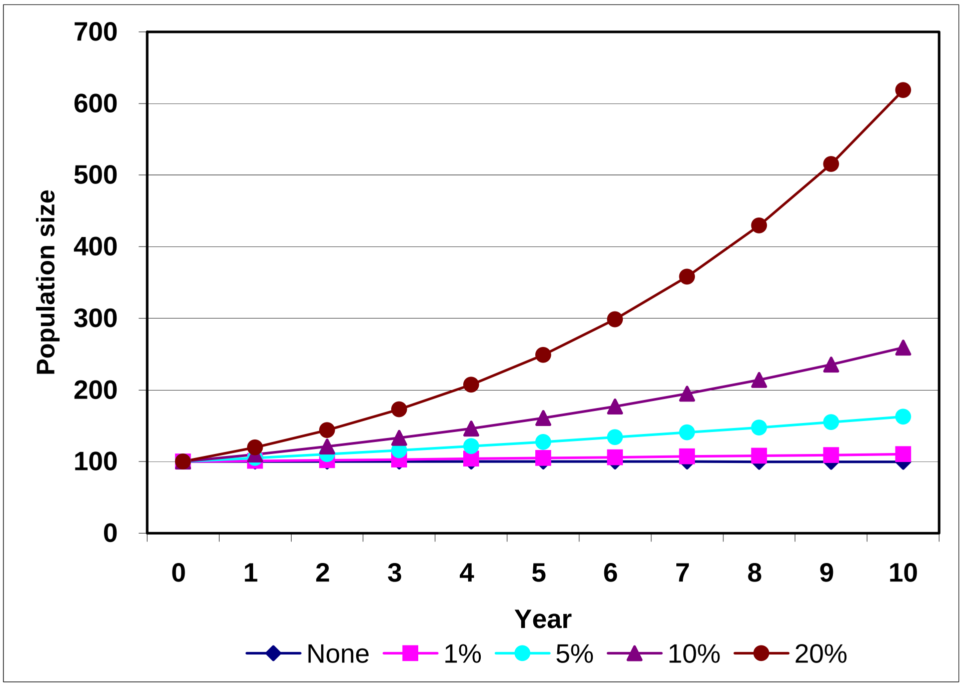
The ecological assumptions underlying the assessment are also important. These were examined as far as possible using the data that are now available on the short-term population biology of rubber vine. Most important are recruitment, growth and mortality rates and their responses to fire. These processes are critical because they will determine the frequency with which fire is required.

Ecological assessment

As indicated under Objective 1 in this report, seedlings of rubber vine can be regarded as established at 2-4 years of age but probably take 5-10 years to reach reproductive maturity. Neither of these figures indicate likely rates of population increase. This would require long-term and intense monitoring of recruitment rates. To make up for this hiatus in our understanding of rubber vine population dynamics for the sake of reviewing the economics of using fire as a control method, we have examined the outcomes of imposing fire on hypothetical populations of rubber vine that differed in the recruitment rates that they exhibited.

Longer-term responses to fire were simulated using a population projection model that was based on growth and mortality data collected during short-term field studies conducted at Lansdown (see Objective 3). Growth and mortality patterns over 12 month periods were expressed as transition matrices that could be used to simulate changes in population size and height class structure over a hypothetical 10 year period under various scenarios. In particular, we examined the sensitivity of population growth to fire in populations with recruitment rates that range from 0% to 20% per annum. Seedling mortality data for rubber vine (see Objective 1) suggest that recruitment rates are likely to be in the lower part of this range even for populations that are growing in a relatively mesic environment. Population growth was simulated with recruitment rates of 0%, 1%, 5%, 10% and 20% p.a. combined with either no fire or a single fire in the second year of a 10 year period (Figure 30). Initial populations in each case were the same, being dominated by small plants and so simulating the early stages of an infestation.

(a)



(b)

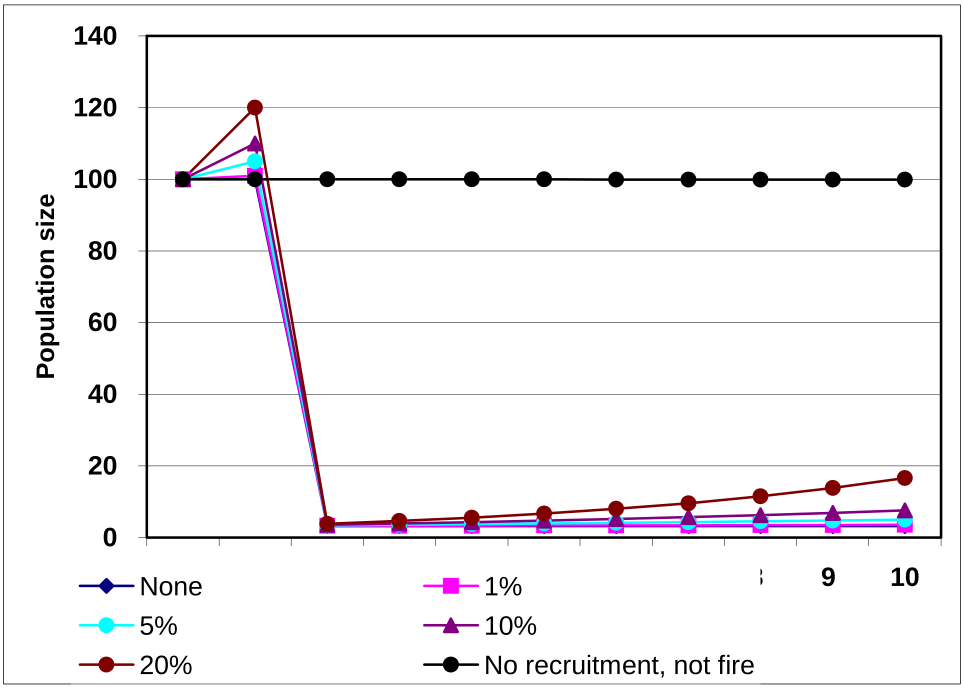


Figure 30. Simulated changes in the size of rubber vine populations subjected to either (a) no fires or (b) a single fire in the second year of the simulation. In each case, simulations with no recruitment or recruitment rates calculated at 1%, 5%, 10% and 20% of the population were examined. Initial populations were dominated by pre-reproductive individuals. Populations changes with no fire and no recruitment are shown for comparison in part (b).

This analysis suggests that eight years after a single fire the rubber vine population would not recover to 20% of its pre-fire size even if the recruitment rate was as high as 20% per annum

(Figure 30a). In contrast, unburnt populations increased over the simulated time periods, more than doubling under a scenario in which the populations increased at 10% per annum (Figure 30b).

When the scenarios involved initial populations dominated by individuals that were of reproductive size, those populations did not reach pre-burn densities eight years after the fire (Figure 31). Even 20% recruitment, which is probably much higher than is likely in reality, leads to only 70% of the pre-fire population eight years after fire.

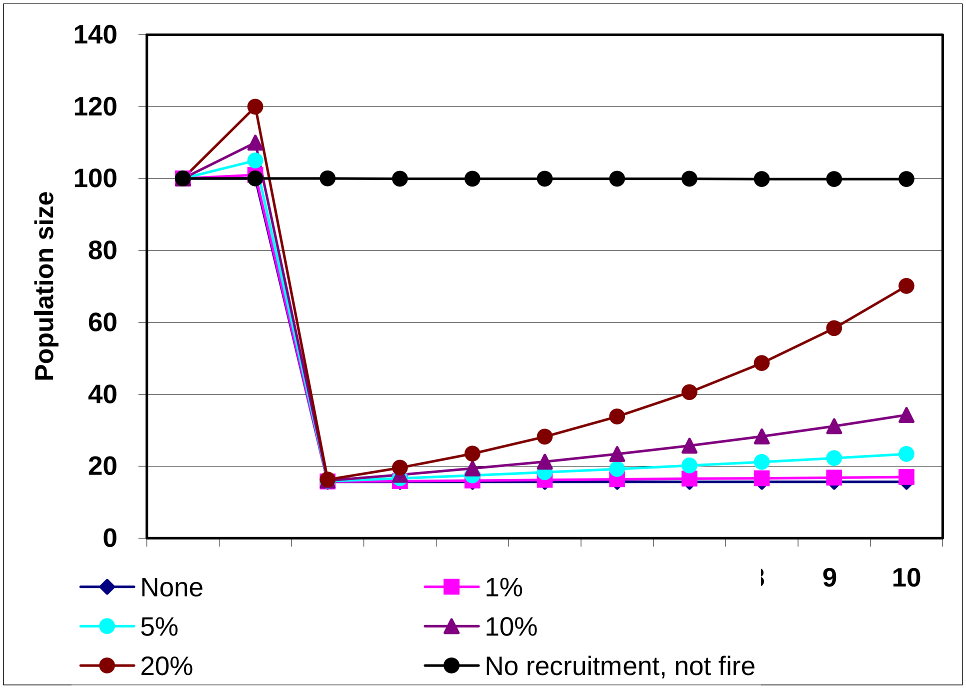


Figure 31. Simulated changes in the size of burnt rubber vine populations with no recruitment or recruitment rates calculated at 1%, 5%, 10% and 20% of the population. Initial populations were dominated by pre-reproductive individuals. Populations changes with no fire and no recruitment are shown for comparison.

Other issues relating to the integration of economic and ecological information in the management of rubber vine

At least two other ecological factors are likely to have a major impact on the economics of rubber vine management: fuel dynamics and the impact of biological control agents. Moreover, these two aspects probably have an important bearing on one another.

Capacity for effective burning for rubber vine management is governed in large measure by fuel availability. Fuel availability varies greatly between years and the likelihood of adequate fuel varies across the climatic range occupied by rubber vine. Incorporating prescribed fire into a management programme for rubber vine will be more readily achieved in those parts of the species' range with higher average annual rainfall and more reliable rainfall. In drier areas it may be most appropriate to limit burning to those years that have above-average rainfall. This will minimise the opportunity costs that result from destocking.

The proliferation of two defoliating biological control agents is important in a number of ways. First repeated defoliation, while it may not kill large numbers of established plants, does facilitate higher herbaceous production. This contributes to fuel loads and fuel continuity and so encourages the use of prescribed burning as a control technique. Second, while there are few data available, the combined effect of the biological control agents on the reproductive output of rubber vine is apparently significant. Anecdotal evidence suggests that total pod production by rubber vine has greatly diminished since the rust and insect agents have become prevalent. This has major implications for the fire regime required to effect control. The economic analysis assumed infestations should be burnt every eight years. The impact of biological control agents raises the prospect that the frequency of burning could be decreased greatly once the initial infestation has been reduced. The rate of reinfestation will be much lower where the biological control agents decrease seed production.

A final issue relates to the landscape-scale patterns of rubber vine infestation. It is difficult to take into account, in an economic assessment, the fact that rubber vine is unevenly distributed across the landscape and, moreover, that this heterogeneity is not stochastic, but is very strongly influenced by habitat. Rubber vine infestations are usually concentrated in the more mesic parts of the landscape. This is important because these parts of the landscape are usually the most productive but also because effective prescribed burning may be more ecologically feasible there due to the prospect of higher fuel loads. It also means that useful prescribed fires can be restricted to parts of the landscape. This will alter the economic feasibility of a burning strategy.

REFERENCES

Brown, J.R. and Ash, A.J. (1996) Pastures for prosperity. 4. Managing resources: moving from sustainable yield to sustainability in tropical rangelands. Tropical Grasslands 30:47-57.

Brown, J.R. and Carter, J. (1998). Spatial and temporal patterns of exotic shrub invasion in an Australian tropical grassland. *Landscape Ecology* **13**: 93-102.

Burrows, W.H., Carter, J.O., Scanlan, J.C., and Anderson, E.R. 1990. Management of savannas for livestock production in north-east Australia: contrasts across the tree-grass continuum. *Journal of Biogeography* 17:503-12.

Carter, J.O. (1994). *Acacia nilotica*: a tree legume out of control. *In* 'Forage tree legumes in tropical agriculture', eds. R.C. Gutteridge and H.M. Shelton, pp.338-51. (CAB International, Wallingford, U.K.).

Cobon, D.H. (1994). Developing practical biological control strategies for gidyea and prickly acacia. Final Report, DAQ 58, AWRAP, Queensland Department of Primary Industries.

Cousens, R. 1987. Theory and reality of weed control thresholds. *Plant Protection Quarterly* 2:13-20.

Kriticos, D.J., Brown, J.R., Maywald, G.F., Sutherst, R.W., and Adkins, S.A. 1999. A population model of Acacia nilotica: A tool for exploring weed management and the effects of climate change. Proceedings of the VI International Rangelands Congress; 1999 Jul 17-1999 Jul 23; Townsville, Australia.

Kriticos, D.J., Brown, J.R., Radford, I.D., and Nicholas, M. submitted. Plant Population Ecology and Biological Control: *Acacia nilotica* as a Case Study. *Biological Control* .

Mackey, A.P. (1997). The biology of Australian weeds 29. *Acacia nilotica* ssp. *indica* (Benth.) Brenan. *Plant Protection Quarterly* **12**, 7-17.

Mackey, A.P. (1998). *Acacia nilotica* ssp. *indica* (Benth.) Brenan. *In* 'The Biology of Australian Weeds' (eds, F.D Panetta, R.H. Groves and R.C.H. Shepherd) Vol. 2, pp. 1-18. (R.G. and F.J Richardson, Melbourne).

March, N. Queensland Department of Natural Resources (then Department of Lands) (1995). Exotic Woody Weeds and their Control in North Western Queensland. Eds. N. March. (Isa Printing Service, Mt Isa).

Mooy, L.M., Scanlan, J.C., Bolton, M.P., and Dorney, W.J. 1992. Management guidelines derived from ecological studies of prickly acacia (*Acacia nilotica*). Proceedings of the 1st International Weed Control Congress; 1992 Feb 17-1992 Feb 21; Monash University Melbourne, Australia. International Weed Science Society and Weed Science Society of Victoria. pp 347-9.

Pratt, D.J. and Knight, J. (1971). Bush-control studies in the drier areas of Kenya V. Effects of controlled burning and grazing management on *Tarchonanthus/Acacia* thicket. *Journal of Applied Ecology* **8**: 217-237.

Radford, I.J., Kriticos, D., Nicholas, M. and Brown, J.R. (1999). Towards an integrated approach to the management of *Acacia nilotica* in Northern Australia. *Proceedings of the International Rangeland Congress, Townsville.*

Radford, I.J., Brown, J.R., Nicholas, M. and Kriticos, D. (unpublished). Habitat specific seed production and dispersal patterns of *Acacia nilotica* ssp. *indica* (Benth.) Brenan (prickly acacia), an invasive shrub of northern Australia.

Thompson, K.T. (1992). *Prickly acacia infestation of the Mitchell grasslands*. Final Report: National Soil Conservation Programme. Richmond Landcare Group.

COMMUNICATION OF RESULTS

Over the last four years, there have been some very significant developments in the management of exotic woody weeds in northern Australia. These include the release of the National Weeds Strategy in 1997, the subsequent recognition of Weeds of National Significance (WONS), and the development of national strategies for individual northern shrub weeds including prickly acacia and rubber vine. It addition, there has been the establishment, spread and significant impact of two biological control agents for rubber vine and the increasing acceptance of fire as a key tool in rubber vine management. These developments cannot, of course, be solely attributed to work conducted under the objectives of this project or its predecessor MRC Project CS219. However, both of these projects have made a major contribution to the development of effective strategies for the management of three WONS, rubber vine, prickly acacia and mesquite as well as chinee apple, another species that is recognised as being nationally important even though it does not have WONS status. It has been vital that communication of the results of this work take place in conjunction with these other developments relating to exotic woody weeds in the north.

Personnel involved (CSIRO Tropical Agriculture and Queensland Department of Natural Resources Tropical Weeds Research Centre) in or having close links with (Queensland Department of Natural Resources Tropical Weeds Research Centre) this project have been key players in the development of the National Strategies for prickly acacia and rubber vine. They have contributed as members of the National Rubber vine and Prickly Acacia Steering Group and the Rubber vine and Prickly Acacia Management Groups. This has provided opportunities for the results of this project to be disseminated to pivotal stakeholders who are playing active roles in the co-ordination and promotion of exotic woody weed management in northern Australia.

The results of this project have been communicated in a variety of ways, supporting national strategies where they exist. The results have been verbally communicated, most notably through interaction with various Landcare groups including Etheridge, Dalrymple, Pentland, Seventy Mile Range, Lynd, and Bowen. There has been continuing involvement with the activities of several of these groups on weed issues, notably with the Seventy Mile Range Landcare Group.

Large scale demonstrations of the effectiveness of fire against rubber vine have been conducted on Wrotham Park Station in the Mitchell River district. This involved two fires in successive years on a 16 square km site and achieved approximately 90% mortality in areas of very dense rubber vine. The success of this exercise prompted initiation of a far more extensive programme of riparian burning on the Walsh and Mitchell Rivers by the management of Wrotham Park Station. The demonstration has also attracted considerable attention well beyond the district. Other demonstrations of fire for rubber vine control have been conducted at Charters Towers, Georgetown, Home Hill and Rockhampton.

Similar extensive demonstrations of prescribed burning have been conducted against mesquite (*Prosopis pallida*) in the Mitchell grass country around Richmond and Hughenden.

Results of this work were also presented in the form of principles of weed management at the very successful Meat Profit Days conducted at Emerald in 1998 and Charters Towers 1999.

Conference and Workshop Presentations and Publications

Campbell, S.D. and Grice, A.C. 1998 Fire management for exotic plant control in northern Australia. 7th Queensland Fire Workshop, Cairns.

Campbell, S.D. and Setter, C.L. 1999 Mortality of *Prosopis pallida* following burning. Proceedings of VI International Rangeland Congress.

Grice, A.C., Walker, L. and Whiteman, L. 1999 An invasive rangeland shrub responds rapidly to top removal. Proceedings of the VI International Rangeland Congress. Townsville, Australia.

Grice, A.C. and Brown, J.R. 1999 Invasive shrubs in Australian tropical rangelands: past, present and future research. Proceedings of the VI International Rangeland Congress. Townsville, Australia.

Keir, M.J., Campbell, S.D., Lindsay, A.M. and Setter, C.L. 1999 Seedling emergence in mesquite infestations. Proceedings VI International Rangeland Congress, Townsville, Australia.

Kriticos, D.J., Brown, J.R., Maywald, G.F., Sutherst, R.W., and Adkins, S.A. 1999. A population model of Acacia nilotica: A tool for exploring weed management and the effects of climate change. Proceedings of the VI International Rangelands Congress. Townsville, Australia.

McKenzie, J.R. 1999 Rubber vine Control – Fire. Northern Australia Fire Workshop, Cooktown.

Radford, I.J., Kriticos, D., Nicholas, M. and Brown, J.R. (1999). Towards an integrated approach to the management of *Acacia nilotica* in Northern Australia. *Proceedings of the International Rangeland Congress, Townsville.*

Scientific Journal Publications

Grice, A.C. 1998 Ecology in the management of Indian jujbe (*Ziziphus mauritiana*). Weed Science 46:467-474.

Kriticos, D.J., Brown, J.R., Radford, I.D., and Nicholas, M. submitted. Plant Population Ecology and Biological Control: *Acacia nilotica* as a Case Study. *Biological Control* .

Media Releases

Press release “Mesquite to the Torch”. Northern Muster Newsletter, North Queensland Register and Queensland Country Life. March 1999.

Press release “Rubbervine Control – Fire”. Queensland Country Life. May 1999.
Press release “Rubbervine Reaps Killer Results”. Savanna Links Newsletter. January 1999.

Press Release “Fire puts heat on rubber vine”. Queensland Country Life. March 1999.