The sustainability of a commercial harvest of the Arnhem Land cycad (*Cycas arnhemica*) by an Aboriginal community: Impacts on Growth, Survival and Recruitment

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Statement of authorship

I declare that this thesis is my own work and has not been submitted in any form for any other degree or diploma at any other university or other institute of tertiary education. Information derived from the published and unpublished work of others has been acknowledged in the text and list of references.

Julia Schult

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List of Abbreviations

AIC	Akaike's Information Criterion
ANOVA	Analysis of variance
BAC	Bawinanga Aboriginal Corporation
CDEP	Community Development Employment Program
CITES	Convention on the International Trade of Endangered Species of Wild Flora and Fauna
DF	Degrees of freedom
ENRC	Environment and Natural Resources Committee
IUCN	World Conservation Union
NT	Northern Territory
PWCNT	Parks and Wildlife Commission of the Northern Territory
QAICc	Akaike's Information Criterion with quasi-likelihood modification
SE	Standard error

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Abstract

The commercial harvest of the endemic Arnhem Land cycad, *Cycas arnhemica,* provides an opportunity for the local Aboriginal community to establish a small-scale community-based enterprise. *C. arnhemica* is protected under Northern Territory and Commonwealth legislation and permits for commercial harvest are conditional upon monitoring of the population to ensure the sustainability of the harvest. An experimental harvesting and monitoring program was established by the Key Centre for Tropical Wildlife Management and Bawinanga Aboriginal Corporation (BAC) in 2001 to determine the effects of harvesting on growth, survival and recruitment of *C. arnhemica.* Monitoring was carried out in replicate harvest treatments using tagged individuals. The survival of harvested plants under a range of fungicide and rooting hormone treatments in the nursery was also investigated to minimise post-harvest loss of plants. A range of *a priori* candidate models was tested to determine which factors influenced cycad life history using information-theoretic methods. Harvesting was not found to have an effect on life history traits of the population. Survival, recruitment and growth did not differ significantly between harvested and unharvested populations. Survival of *C. arnhemica* in the nursery was not affected by hormone and fungicide treatment compared to untreated control plants but long-term storage of plants without soil led to high mortality. The harvest of *C. arnhemica* at the current rate of 500 individuals per year appears to be sustainable. Recommendations are made to ensure the continued sustainability of the harvest.

1 Introduction

1.1 Commercial use of wildlife

The commercial use of wildlife is a controversial issue (e.g. Diekman 1995). Although few people question the rights of Indigenous peoples to harvest resources for subsistence purposes, the commercial exploitation of those resources is more contentious. Some see the exploitation of wildlife for profit as a threat to biodiversity (e.g. Hoyt 1994; Struhsaker 1998), while others argue that the use of wildlife creates incentives for its protection and helps in the conservation of habitats and endangered species (e.g. Cunningham 2001; Godoy and Bawa 1993; Medellin 1999). To safeguard populations from depletion and potential extinction through over-exploitation, the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) was established in 1973 by the World Conservation Union (IUCN) to ensure the harvest and trade of endangered species does not threaten their survival (CITES 2003). Many questions, ranging from ownership of resources to the protection of wildlife and specific restrictions on trade of wildlife products still need to be resolved (Webb 1997). There is, however, some consensus that, if managed carefully, consumptive and commercial use of wildlife can promote the conservation of wild species (Freese 1998).

In Australia, the harvest of native plants makes a substantial contribution to the economy. For instance, in Victoria 70,000 tree ferns (*Dicksonia antarctica*) with an export value of more than $1 million were harvested in 1996 and 15% of the wildflower production in Western Australia came from wild harvest in 1995-96 (ENRC 2000). The value of Aboriginal sculptures carved from native *Bombax ceiba* wood in Maningrida was estimated at about $200,000 between 1998 and 2001 (Griffiths*et al.* 2003).

Many organisations see wildlife use as a development opportunity for indigenous peoples around the world to derive income from their land (e.g. Australasian Wildlife Management Society 2000; Pittock 1995). The sustainable use of wildlife is encouraged by the World Conservation Union (IUCN 1995), the Australian Government (e.g. Senate Rural Affairs and Regional Affairs and Transport Reference Committee 1998) and the Parks and Wildlife Commission of the Northern Territory (PWCNT 1995). In its "Strategy for Conservation through the Sustainable Use of Wildlife" the PWCNT explicitly states as one of its objectives *"to ensure that*

Aboriginal people (...) have the option to develop commercial uses [of wildlife] on a sustainable basis" (PWCNT 1995, p.1).

1.2 Aboriginal economic development

Aboriginal people in Australia today are economically disadvantaged. The economic status of Aboriginal people in relation to employment, income, housing, education and health is the lowest of all Australians and the state plays a large role as a source of income for many families and as a provider of services for Aboriginal people (Altman 2000; Taylor 2003). There are few opportunities in remote Aboriginal communities to establish commercial ventures to reduce this dependency on the state (Altman 2001).

Previous attempts to create market opportunities and establish commercial enterprise in Aboriginal communities have shown that externally driven projects have failed in many cases and that a community-based approach is needed if these commercial ventures are to succeed (Dale 1996). A lack of control by Aboriginal people and a lack of respect for local traditions and custom are amongst the reasons that have lead to project failure in the past (Bomford and Caughley 1996; Hughes 1996). The contemporary Aboriginal art industry shows that community-based enterprise can be successful when it is utilising local resources and is controlled by Indigenous people. The Aboriginal art industry today contributes $100-300 million per year to the Australian economy (Altman 2002), and much of it relies on native species for weaving, carving and bark painting.

Many Aboriginal people have expressed the desire to become involved in the sustainable commercial use of wildlife, because it builds on traditional knowledge and skills and can provide income and employment that is more culturally appropriate than many other commercial activities (Cleary 1995; Senate Rural Affairs and Regional Affairs and Transport Reference Committee 1998). Therefore, the commercial harvest of wildlife may provide an opportunity for community-based enterprise through an expansion of the traditional customary economy into sustainable commercial use (Freese 1998).

1.3 Sustainable use of wildlife in Maningrida, Arnhem Land

The Maningrida region of central Arnhem Land has a limited economic base. The Maningrida township was established in 1950 and today services more than 2000 people. Employment is largely limited to regional services, some not-for profit organisations and the Community Development Employment Program (CDEP), which includes activities like the delivery of municipal services, employment with service and infrastructure providers, social and cultural services, office administration as well as the manufacture of art and craft on outstations (Bawinanga Aboriginal Corporation 2001). The establishment of a cypress pine forestry project in the 1960s created little employment in the Maningrida community and eventually failed (Hughes 1996). The most significant exports of the region today are cultural products sold via Maningrida Arts and Crafts (Altman and Taylor 1989). Sustainable use projects like the commercial harvest of crocodiles and crocodile eggs and the sale of turtle hatchlings for the pet trade are providing small-scale commercial opportunities for local Aboriginal people (Bawinanga Aboriginal Corporation 2001). Benefits of such projects include the development of strategies for sustainable economic growth in the community as well as the preservation of traditional knowledge and the cultural survival of traditional people (Cotton 1996; Vardon*et al.* 1997). In 2001 the Bawinanga Aboriginal Corporation obtained a permit to commence the commercial harvest of a local endemic cycad species, *Cycas arnhemica*, from woodland areas near one of the outstations.

1.4 Cycads - Biology and Threats

Cycads are long-lived, dioecious, woody plants with a palm-like growth habit. Their distribution ranges from Africa and Central and South America to Southeast Asia and Australia (Jones 1993). Cycads are slow growing and reproduction occurs from seed or vegetative sprouting (Watkinson and Powell 1997). They are insect-pollinated plants and the vectors responsible for pollination are usually weevils that are closely dependent on the cycads (Hill 1998). Reproduction is often sporadic and not all mature individuals reproduce every year (Jones 1993). Cycads live in symbiosis with nitrogen-fixing cyanobacteria that are contained in coralloid roots and enable the plants to grow on nitrogen-poor soils (Medeiros and Stevenson 1998). Cycads possess tuberous roots that enable them to store nutrients and allow regrowth from old root stocks when the above-ground foliage and sometimes the stem has been destroyed by fire (Hill 1998). They are fire-tolerant and are often the first plants to produce new leaves after a fire, making them a spectacular sight in the burnt landscape. Accurate ageing of cycads is difficult because juveniles often remain at ground-level for several years. They grow an underground stem before the emergence of an above-ground stem and have the ability to re-sprout from old root stock.

Worldwide, cycads are under threat from habitat destruction and collection (Donaldson 2003). Many species are rare and a number of species are listed by the World Conservation Union (IUCN) as endangered or critically endangered (Donaldson 2003). In Australia, however, most species are locally common and less vulnerable to extinction than cycads in other parts of the world (Donaldson 2003).

1.4.1 Cycad management in the Northern Territory

There are twelve species of cycad in the Northern Territory, most of which are endemic with no significant populations outside the Northern Territory. Only one of these species, *Macrozamia macdonnellii*, is considered vulnerable to extinction (PWCNT 1997). All Northern Territory cycads are listed in Appendix II of the *Convention on International Trade of Endangered Species of Wild Flora and Fauna* (CITES). Species listed in this appendix are not necessarily considered endangered but can only be traded internationally if an approved management plan is in place to minimise impacts on wild populations. As a signatory to the convention Australia is obliged to meet these requirements, which are effected through the *Environmental Protection and Biodiversity Conservation Act* 1999 and the *Territory Parks and Wildlife Conservation Act* 2000.

Until the 1980s the Department of Agriculture encouraged pastoralists to eradicate cycads because they are poisonous to cattle (Wesley-Smith 1980). In 1997 a trial management program was formulated by the Parks and Wildlife Commission of the Northern Territory (PWCNT 1997) to ensure the conservation of Northern Territory cycad populations. The management program aims to maintain viable populations of all cycad taxa and habitats across their range. Its objectives are to develop strategies for the sustainable use of cycads and to implement a system of monitoring the impacts of harvesting on cycad populations as well as to encourage research into cycad ecology (PWCNT 1997). The management program specifically allows for the commercial harvest of cycads. Originally, commercial harvesting was limited to a maximum of 500 stemmed plants per year but the program was revised in 2003 and harvest limits were adjusted to an annual offtake of up to 5% of the population (PWCNT 2003). A permit from the PWCNT is required to take any cycads for commercial use and the actual harvest limit is subject to the director's discretion.

1.4.2 Cycads as a commercial opportunity

The Arnhem Land Cycad (*Cycas arnhemica*) is endemic to an area between the Blyth and Glyde Rivers in central Arnhem Land in the wider Maningrida region. *Cycas arnhemica* is abundant throughout its range (Hill and Osborne 2001) and a recent preliminary population estimate of an area of 86 km² around Gamardi outstation suggests a population size of almost 5 million individuals, not including another 200-300 km² of inaccessible *C. arnhemica* habitat (Griffiths et al., unpublished data). Currently, the conservation status of *C. arnhemica* on the IUCN Red List is classified as being of "least concern". This classification was allocated despite assuming a population size of only 10,000 individuals (Donaldson 2003).

Cycas arnhemica has a columnar growth form and looks similar to *C. armstrongii*, a cycad that is common in the understorey of savanna vegetation around Darwin. Cycads are popular ornamental plants and are highly sought after by cycad enthusiasts, collectors, botanic gardens and horticulturalists. *C. arnhemica* is currently not available for purchase from anywhere in the world and therefore provides a unique commercial opportunity for Aboriginal traditional owners.

1.5 Monitoring of cycad harvest

As a CITES listed species, there is a requirement for monitoring of any extraction of *Cycas arnhemica* plants for commercial purposes. Information about cycad ecology and population dynamics is limited and only few studies have investigated the impact of whole plant collection on cycad populations. One of these studies, using simulation modelling of harvesting impacts on two African cycad species, suggests that the harvest of adult plants would have a profound negative effect on cycad populations and seedling recruitment (Raimondo and Donaldson 2003).

The management plan recognises that there is a lack of information about the ecology of Northern Territory cycads. However, it is government policy not to use the lack of detailed information about a species as an argument against commencement of commercialisation on a trial basis (ENRC 2000; Senate Rural Affairs and Regional Affairs and Transport Reference Committee 1998). The Northern Territory cycad management plan places the emphasis on monitoring of harvested populations and adaptive management where salvage harvest of cycads from land that is to be cleared for other land uses is not possible.

In 2001 experimental harvests were conducted by the Bawinanga Aboriginal Corporation in association with the Key Centre for Tropical Wildlife Management at Charles Darwin University to assess the commercial feasibility and ecological sustainability of a cycad harvest. A monitoring program was set up in accordance with the PWCNT cycad management plan to assess the impacts of harvesting of different life stages on the ecology of *C. arnhemica*. Additionally, harvested cycads were subjected to a range of nursery treatments to identify the most efficient growing regime. The objective of this nursery trial was to minimise mortality of cycads after harvest to reduce wastage of plants.

1.6 Aims

The aim of this Masters project was to evaluate the monitoring program and to assess the effects of different experimental harvesting regimes on *C. arnhemica* using data collected as part of the monitoring program from 2001 to 2003, and to determine the effectiveness of various nursery treatments to improve survival of harvested plants. Specifically, the aims of this study were to:

1. Determine the impact of experimental harvest on survival of *in situ Cycas arnhemica.*

2. Determine the impact of experimental harvest on recruitment of *in situ Cycas arnhemica.*

3. Determine the impact of experimental harvest on stem growth of *in situ Cycas arnhemica.*

4. Determine survival of *Cycas arnhemica* in a range of post-harvest nursery establishment trials.

5. Make recommendations for future cycad management.

2 Methodology

2.1 Study Area

The study sites for the experimental harvest were located near Gamardi outstation on the Blyth River, 50 km southeast of Maningrida in north-central Arnhem Land, at 12° 16'S, 134° 41' E (Figure 1). The region has a wet-dry tropical climate with a long winter dry season from May to November and a summer wet season from December to April. Mean annual rainfall at Maningrida is 1268 mm and mean temperatures range from 17°C – 30°C in the dry season to 25°C -33°C in the wet season. Relative humidity is high throughout the year and mean values at 9 am range from 70% in the dry season to 84% during the wet (Bureau of Meteorology 2003). The sandy soils of the region support open forest dominated by *Eucalyptus tetrodonta* and *E. miniata* with low perennial tussock grasses in the understorey. *Cycas arnhemica* is locally abundant in the forest midstorey (Hill and Osborne 2001)

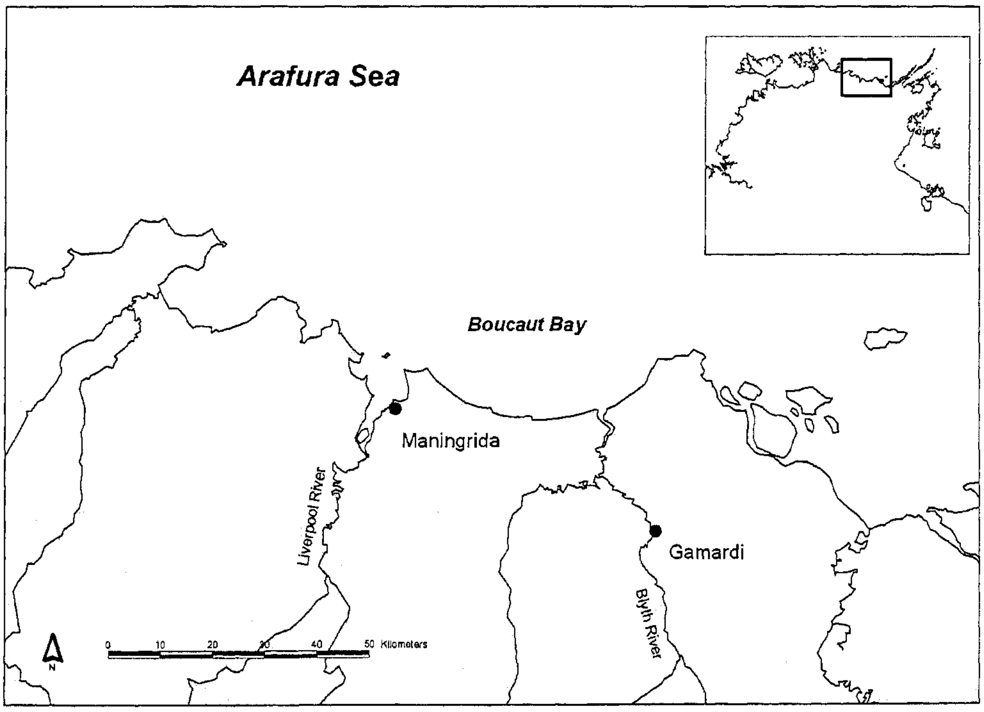

Figure 1. Map of the Maningrida region of central Arnhem Land, showing the location of Gamardi outstation, where study sites for *Cycas arnhemica* harvest were situated.

2.2 Study Design

The monitoring program was established in the Gamardi area in August and September 2001. Three sites were randomly chosen and at each of these sites nine 20 m x 20 m quadrats were marked. Three replicates of the following harvest treatments were then randomly assigned to the quadrats and all suitable plants of the assigned size class were removed:

1. Harvest of adult cycads (0.5 – 1.0 m)

2. Harvest of juvenile cycads (0.05 m – < 0.5 m)

3. No harvest

2.3 Data Collection

2.3.1 Survival, growth and recruitment

Data on cycad survival and stem growth were collected annually for two consecutive years. All cycads within each quadrat were individually tagged in July 2001, and the height and diameter of all plants were measured to the nearest centimetre and re-measured in September 2002 and July 2003 (Julia Schult), using a measuring pole and callipers. In July 2003 plant status was recorded as alive or dead according to the presence of leaves and stem condition. Plants were scored as dead where stems had collapsed and there was no evidence of regrowth. Plants that had been tagged previously but could not be found in subsequent years were also considered dead so as to obtain a conservative estimate of survival. The height and diameter of all tagged plants were measured. Height was measured as the length of the stem from the ground to the point of emergence of the oldest leaves. To assess recruitment into the quadrats, a 5 m x 20 m strip in each quadrat was searched for new seedlings in 2003 and the number of seedlings in the area was recorded. For each quadrat the occurrence of fires was recorded annually and an assessment of ground disturbance was carried out in July 2001 and July 2003.

2.3.1.1 Limitations

Community involvement was a priority for all aspects of the project so that data collection was performed by a many different people. This led to frequent measurement errors. Where measurements showed obvious sampling errors, i.e. shrinkage, or growth of >15 cm/year, the data were excluded from the analysis.

2.3.2 Survival of harvested cycads

Cycads were harvested from quadrats in 2001 by digging around the stem to a sufficient depth to reveal the root bulb. Taproots were severed below the main part of the bulb and care was taken to preserve as much root material as possible. The harvested cycads were stored in shady conditions until they were transferred to the BAC nursery in the Maningrida township for horticultural treatment. At the nursery, each plant was randomly assigned one of five nursery treatments prior to potting (Table 1).

Table 1: Nursery treatments applied to harvested *C.arnhemica* prior to potting. Sample sizes are given for juvenile and adult size classes.

Treatment	Sample Size	
	Adult (0.5-1.0 m)	**Juvenile (0.05-< 0.5 m)**
Stored bare-rooted for several weeks	13	32
Root bulb treated with fungicide (domestic chlorine bleach)	11	35
Root bulb treated with rooting hormone	12	29
Root bulb treated with fungicide and rooting hormone	22	49
No treatment	18	29
Total	76	174

For the bare-rooted treatment all soil was removed from the roots of the plants and plants were stored in shady conditions to assess the possibility of exporting plants interstate and overseas. Export regulations do not permit the transporting of unsterilised soil across state and national borders.

For the other treatments, root bulbs were dipped in solutions of diluted bleach or rooting hormone. After application of the respective treatments the plants were potted and kept in the nursery, where they received equal amounts of slow-release fertiliser and were watered regularly. The condition of the harvested cycads was recorded biannually and plants were classified as alive or dead using presence of leaves and general stem condition as indicators of survival.

2.4 Model fitting and selection

2.4.1 Survival *in situ*

An analysis of covariance was carried out to detect differences in survival between harvest treatments. Proportional data often do not conform to the common assumptions of statistical tests, such as constant variance and normal distribution of errors (Crawley 2002). The data were therefore arcsine transformed prior to analysis to allow a linear mixed-effects model to be fitted to the data, which also permitted the use of a combination of fixed and random explanatory variables (Crawley 2002).

Factors considered in model fitting and selection included the fixed effects of harvest treatment, fire frequency and disturbance and the random effects of location. Factors were regarded as fixed when their levels contained informative values (Crawley 2002). The environmental factors fire and disturbance were chosen for their known potential to influence cycad life history (e.g. Negron-Ortiz and Gorchov 2000). Details of the factors and their levels are shown in Table 2.

Table 2: Factors used in model fitting for linear mixed-effects models of *in situ* survival and recruitment of *C.arnhemica*. All factors were specified as categorical variables with harvest treatment, fire and disturbance considered fixed factors due to their informative value and site the only random factor.

Factor	Levels	Notes	Fixed/Random
Harvest Treatment	a: adults j: juveniles c: control	Plants with stems of >50 cm were classified as adults, stems <50cm as juveniles. Treatment indicates which life stage was removed from the quadrat.	Fixed
Fires	0-2	Number of fires that occurred in the quadrat between the harvest in 2001 and sampling in 2003	Fixed
Disturbance	1-5	Scores on a scale of 0-5 for pig, buffalo, human & kangaroo disturbance collected in 2001 and 2003 were added together to give cumulative disturbance score. Disturbance was generally low and cumulative scores ranged from 1-5 only.	Fixed
Site	1-3	Experimental plots were located at 3 randomly chosen sites near Gamardi outstation	Random

Candidate models including both fixed and random effects were constructed as linear mixed-effects models using the software S-Plus, Version 6.1(Insightful 2002). Initially, a null model was constructed and factors were subsequently added to this model to achieve an increase in explanatory power. Instead of the standard restricted maximum likelihood estimation, maximum likelihood was used in the models to allow the comparison of models with different fixed factors (Crawley 2002).

2.4.1.1 Model Selection

The relative explanatory power of the candidate models was assessed by comparing candidate models to the null model using ANOVA comparisons to test the significance of differences between the models. Final model selection from the *a priori* candidate set was based on these comparisons and Akaike's Information Criterion (AIC) (Burnham and Anderson 1998), which ranks models in order of parsimony, penalising the use of additional parameters to achieve a reduction in deviance (Crawley 2002).

2.4.2 Recruitment

Differences in seedling establishment were investigated using analysis of covariance as for *in situ* survival. The recruitment analysis was based on count data, not proportions, therefore a transformation was not necessary. The explanatory variables that influence seedling establishment were considered to be the same as for survival, since fire and disturbance have been known to have an effect on recruitment in other cycads and savanna trees (Liddle 2003; Setterfield 1997)

Candidate models were constructed as linear mixed-effects models and model selection was based on ANOVA comparisons and AIC as above.

2.4.3 Growth

For all individuals with at least two height measurements, the growth rate per year was determined as the difference between the first and last measurement divided by the number of years between these measurements. Stem lengths of zero are often encountered where plants have not yet reached measurable heights above ground because the above ground portion of a cycad stem is the extension of an underground stem. Therefore, cycads that had a stem length of zero at the end of the study period were not included in the analysis because their apparent zero growth rates would have incorrectly skewed the distribution of the data. It was beyond the scope of this study to include measurements of below ground portions of cycad stems.

2.4.3.1 Modelling of non-linear growth

Plant growth is often non-linear and different models have been used to describe plant growth (e.g. Goelz 1999; Liddle 2003). To account for non-linear growth over the lifetime of *C. arnhemica*, a number of growth functions were applied to the data to find the best model to describe the relationship between stem length and growth rate.

The relationship was modelled using non-linear least squares regression. Candidate models included linear, quadratic and exponential growth functions, Hoerl's function for modelling tree growth (Goelz 1999) and a custom function developed by Liddle (2003) for modelling cycad growth.

The equations took the following forms:

Equation 1:	$GR = a\mathrm{SL} + b$	(linear)
Equation 2:	$GR = a\mathrm{SL}^2 + b\mathrm{SL} + c$	(quadratic)
Equation 3:	$GR = ae^{-b\mathrm{SL}}$	(exponential)
Equation 4:	$GR = a\mathrm{SL}e^{-\mathrm{SL}/b}$	(Hoerl)
Equation 5:	$GR = a(\mathrm{SL}+c)e^{-\mathrm{SL}/b}$	(Liddle)

where GR is the annual growth rate, SL is mean stem length over the measuring period (calculated as the mean of initial and final stem lengths), and *a, b* and *c* are constants. The different growth models were fitted to the data for all treatments combined and model fit was compared using AIC differences and mean residual error.

2.4.3.2 Factors influencing growth

Once the model with the best fit had been determined, this model was used in all further analyses. Candidate models examining growth rate as a function of mean height were constructed as non-linear mixed-effect models and included the fixed effects of treatment and fire frequency as well as a random site effect as specified in Table 2. Models with interactions between fixed factors were not considered because the complexity of such non-linear models would have required a larger dataset.

Model selection was based on ANOVA comparisons of candidate models with the null model and AIC. The parameter estimates for different factor levels derived from the preferred models were tested for significant differences using 95% confidence intervals.

2.4.4 Survival in Nursery

Cycad survival under nursery conditions was modelled using binary regression in a binomial generalised linear model.. The analysis was carried out on the binary dataset because individual plant survival was the focus of this analysis and because plant height was included as a continuous explanatory variable. Binary regression is useful when one of the explanatory variables is continuous (Crawley 2002)

Factors that were considered to potentially influence survival of plants in the nursery included post-harvest nursery treatments, plant size class, origin, harvest date and plant height (Table 3).

Table 3: Factors included in binomial generalised linear models of nursery survival of *C.arnhemica*. All factors except height were specified as categorical variables.

Factor	**Levels/Values**	**Notes**
Treatment	BR: stored bare-rooted BL: bleach RH: rooting hormone BLRH: bleach + rooting hormone C: control	Nursery treatments applied after harvesting (see Table 1)
Size Class	j: juvenile (<50 cm) a: adult (>50 cm)	all plants of >50 cm were considered mature
Site	1-3	Experimental plots from which plants were removed
Harvest Date	1: July 2001 2: September 2001	Harvest was carried out in two stages, early and late dry season
Height	0-167cm	Stem height of plants

Candidate models were constructed as binomial generalised linear models (logit link). Initially, probability of survival was modelled as a function of all explanatory factors, giving the following global model:

Probability of Survival ~ Nursery Treatment + Size Class + Origin + Harvest Date + Height

Simplified models were then constructed using backward stepwise model selection to find the most parsimonious model. Candidate models included all single factor models as well as two models considering the additive effects of treatment plus size class and height respectively.

2.4.4.1 Model Selection

The final model selection from the *a priori* candidate set was based on chi-squared tests to determine whether reductions in deviance of candidate models compared to the null model were significant; and on AIC to find the most parsimonious model of those that showed significant reductions in deviance. The relative likelihood of the candidate models was estimated using Akaike weights (Burnham and Anderson 1998).

Due to evidence of overdispersion and the use of a large number of parameters in the global model with a relatively small sample size (n/K ratio 250/23=10.9) quasi-likelihood modification was used with a second-order variant of AIC (QAICc) as recommended by Burnham and Anderson (1998) for n/K ratios of less than 40.

3 Results

3.1 Survival *in situ*

Out of the candidate set of models for cycad survival, no single model was clearly supported. Comparisons of the candidate models with the null model showed no significant increase of explanatory power through the addition of any environmental or harvest treatment effects (p-values = 0.07-0.86, Table 4).

Table 4. Comparison of candidate linear mixed-effects models for *in situ* survival of *C. arnhemica*. ΔAIC shows the difference between the model AIC and the lowest AIC out of the set of models. P-values refer to ANOVA comparison of each model to the null model. Likelihood ratio and ANOVA not applicable for models with no fixed effects.

Model	Factors: Fixed	Factors: Random	AIC	ΔAIC	Likelihood Ratio	p-value (ANOVA with Null model)
Null	-	-	-29.17	0.77	-	-
1	-	Site	-29.30	0.64	N/A	N/A
2	Treatment	Site	-27.07	2.87	1.90	0.39
3	Fires	Site	-29.94	0	4.78	0.09
4	Disturbance	Site	-22.45	7.49	1.28	0.86
5	Treatment + Fires	Site	-29.86	0.08	8.69	0.07
6	Treatment + Fires + Disturbance	Site	-23.63	6.31	10.46	0.23

Mean survival across all harvest treatments over the two-year study period was 0.842. Differences in mean survival were small between harvest treatments with survival ranging from 0.828 (SE=0.018) for adult harvest to 0.835 (SE=0.025) for juvenile harvest and 0.863 (SE=0.023) in the control treatment (Figure 2). The addition of harvest treatment as a factor did not increase the power of the model (p=0.39). Therefore, harvest treatment does not explain the survival pattern of *C. arnhemica* over the two-year study period.

There is a suggestion that fire did influence cycad survival. The models that included fire as a fixed factor had the lowest ΔAICs out of the set of models, however, neither of these models was significantly different from the null model (Table 4). Survival amongst different fire frequencies was lowest in unburnt quadrats and slightly higher in those that had been burnt once or twice in two years but differences in survival between fire frequencies were small (Figure 3).

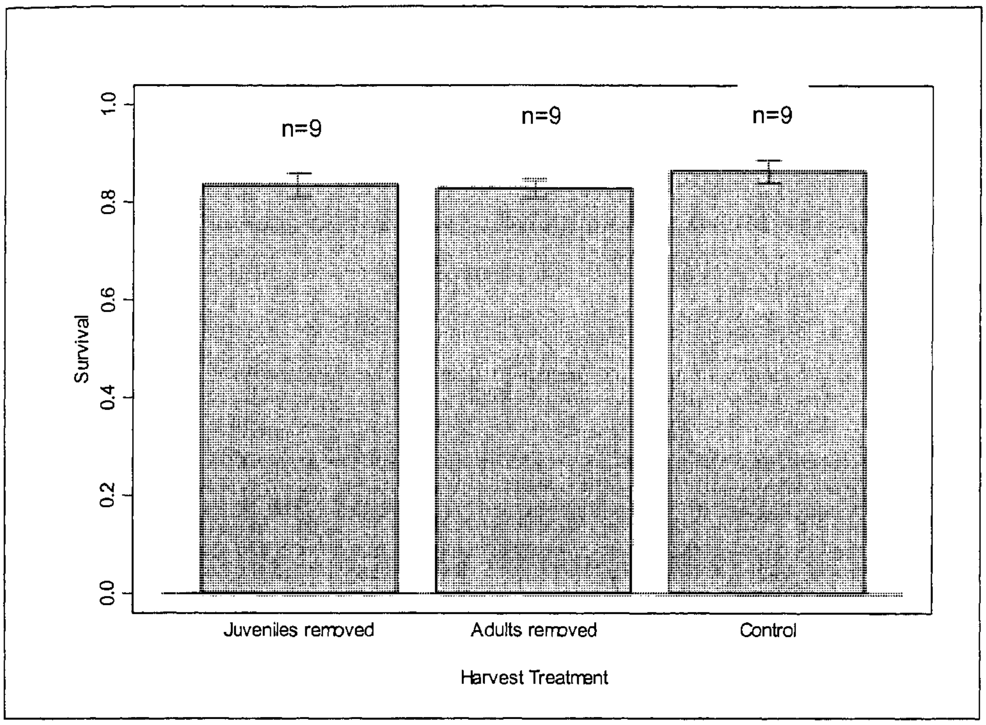

Figure 2. Mean survival of *C.arnhemica* over two years for three harvest treatments. Vertical bars indicate ±1SE.

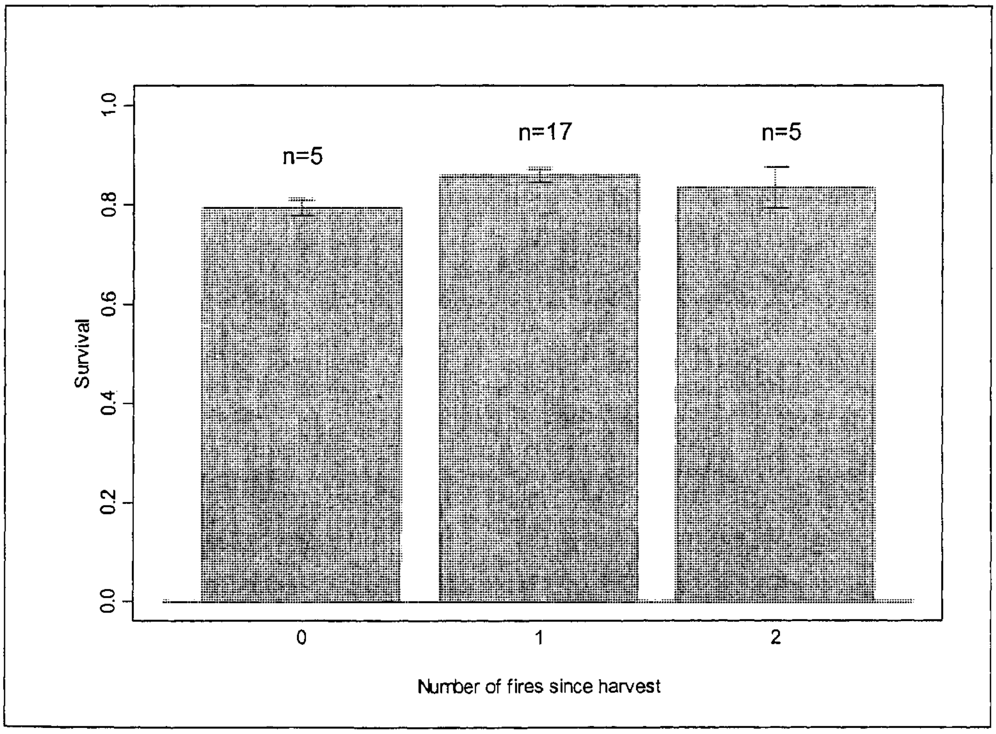

Figure 3. Mean survival of *C. arnhemica* over two years for different fire frequencies. Vertical bars indicate ±1SE.

3.2 Recruitment

No single recruitment model was clearly supported as the best model out of the candidate set. When compared to the null model, none of the candidate models showed a significant increase of explanatory power through the addition of the effects of fire, disturbance or harvest treatment to the null model (p-values = 0.8-0.96, Table 5).

Harvest treatment was one of the candidate models least likely to explain differences in seedling recruitment (p=0.96). The mean number of seedlings was very similar across all harvest treatments and variation between quadrats was high as indicated by large standard errors (Figure 4).

The single-factor model, which contained fire frequency as the only fixed factor, received the lowest AIC. Therefore, fire frequency appears to have some influence on recruitment of new *C. arnhemica* seedlings. However, the support for fire frequency was not strong and it was not significantly different from the null model (p=0.08). Quadrats that had been burnt twice in the two-year study period contained no seedlings while seedling numbers were highest for the intermediate fire frequency and lower for unburnt quadrats (Figure 5).

Table 5. Comparison of candidate linear mixed-effects models for recruitment of *C. arnhemica* seedlings. ΔAIC shows the difference between the model AIC and the lowest AIC out of the set of models. P-values refer to ANOVA comparison of each model to the null model. Likelihood ratio and ANOVA not applicable for models with no fixed effects.

Model	Factors		AIC	ΔAIC	L-ratio	p-value
	Fixed	Random				
Null	-	-	149.59	1.1	-	-
1	-	Site	149.54	1.05	-	-
2	Treatment	Site	153.52	5.03	0.075	0.96
3	Fires	Site	148.49	0	5.10	0.08
4	Disturbance	Site	152.33	3.84	1.26	0.53
5	Treatment + Fires	Site	152.41	3.92	5.18	0.27
6	Treatment + Fires + Disturbance	Site	156.32	7.83	5.27	0.51

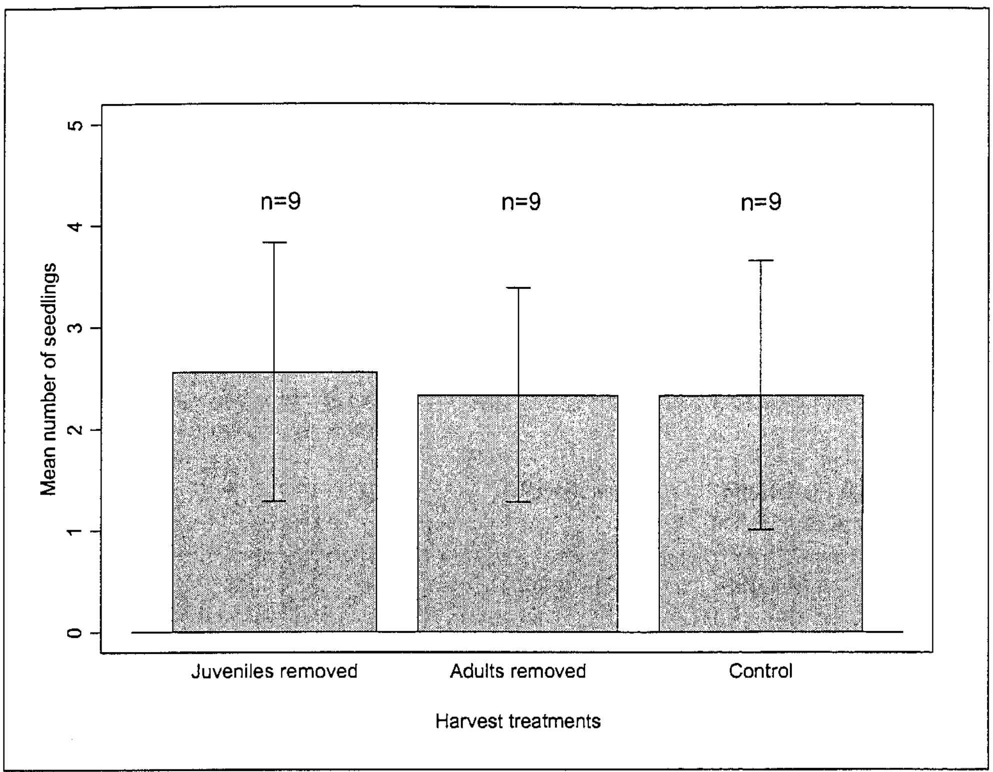

Figure 4: Mean number of new seedlings of *C. arnhemica* for three harvest treatments. Vertical bars indicate ± 1 SE.

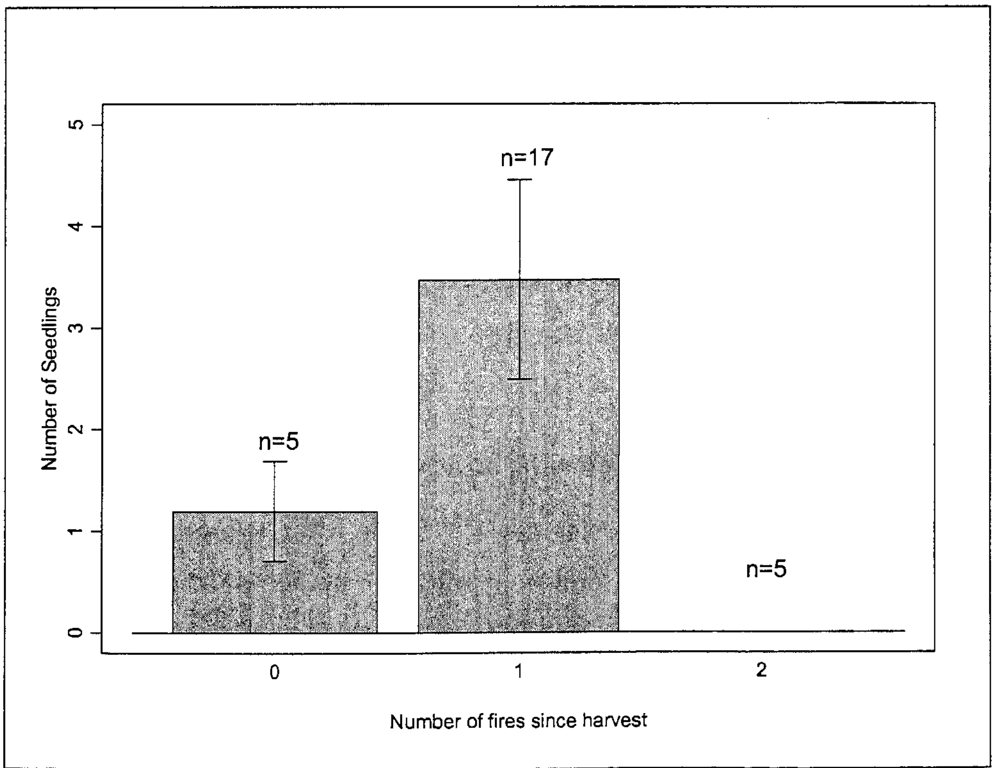

Figure 5. Mean number of *C. arnhemica* seedlings for three different fire frequencies. Vertical bars indicate ±1SE.

3.3 Growth

3.3.1 Suitability of non-linear growth models

The relationship between stem length and growth rate was best represented by Hoerl's function and Liddle's custom function. The Akaike weight indicated stronger support for Hoerl's model and it was therefore adopted as the preferred growth model (Table 6). The remaining four models had essentially no support, with weights that were many orders of magnitude smaller than those for the top two models, and were not considered further. A summary of the full set of models is given in Table 6.

Table 6. Summary of candidate growth models for *C.arnhemica*. Models were constructed as non-linear least squares regression models.

Model	k	ΔAIC	Akaike weight	Residual Sums of Squares
Linear	3	65.77	3.32×10^{-15}	6307.5
Quadratic	4	57.02	2.65×10^{-13}	6107.5
Power	3	47.33	3.36×10^{-11}	5989.2
Exponential	3	67.71	1.26×10^{-15}	6308.0
Hoerl	3	0	0.64	5309.6
Liddle	4	1.12	0.36	5297.8

The mean annual growth rate of *C.arnhemica* for all harvest treatments combined was 3.55 cm/year (n= 393, SE=0.20). Based on Hoerl's function, predicted stem growth rates increase initially with increasing stem length and decrease after a height of approximately 28 cm is reached. At heights of more than 100 cm, growth slows down substantially and the growth rate approaches zero at a stem length of approximately 250 cm (Figure 6).

3.3.2 Impact of harvest treatment on growth

Since Hoerl's model ($GR = aSLe^{-SL/b}$) was selected as the preferred model for cycad growth as a function of stem length, this model was used in non-linear mixed-effects models to assess the effects of harvest treatments and environmental factors on growth. All three candidate models were an improvement over the null model ($p<0.0001$, Table 7). The combination of fire and site effects was selected as the preferred model with a 99.9 % likelihood of being the best model out of the set (Table 7).

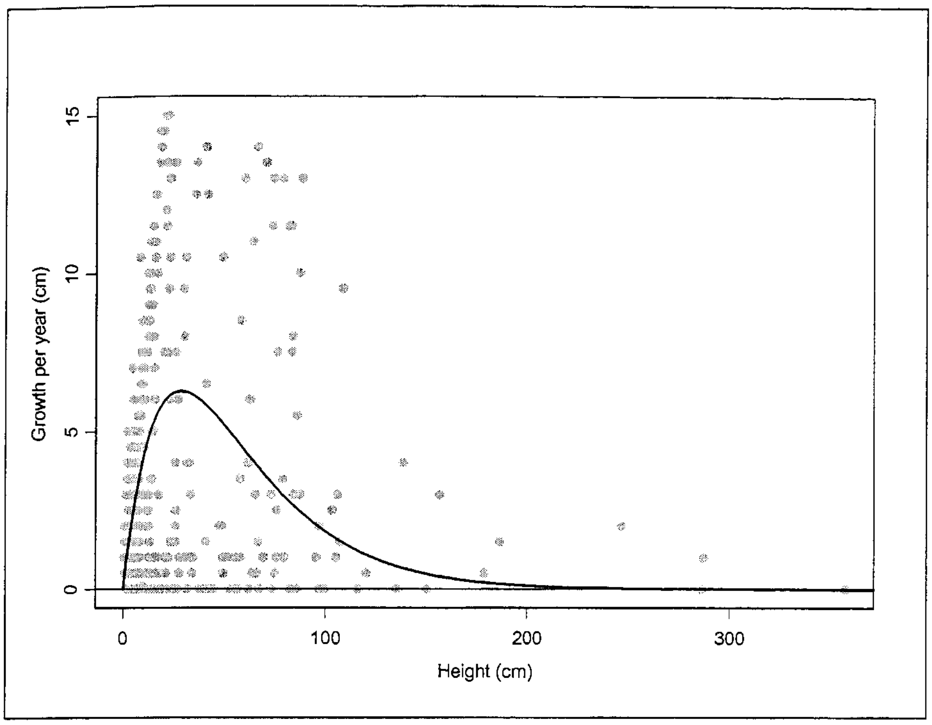

Figure 6. Fitted values for growth rates of *C. arnhemica* as a function of stem length using Hoerl's model. The fitted line is overlaid over observed growth rates.

Table 7. Comparison of candidate non-linear mixed-effects models for growth of *C. arnhemica*. ΔAIC shows the difference between the model AIC and the lowest AIC out of the set of models. P-values refer to ANOVA comparison of each model to the null model. Likelihood ratio and ANOVA not applicable for models without fixed effects. N=363.

Model	Factors		AIC	ΔAIC	Akaike weight	L-ratio	p-value
	Fixed	Random					
Null	-	-	2150	140.5	3.02×10^{-31}	-	-
1	-	Site	2037	27.2	1.23×10^{-6}	-	-
2	Treatment	Site	2024	14.2	0.0008	134.30	<0.0001
3	Fire	Site	2009	0	**0.999**	148.54	<0.0001

None of the other models, including harvest treatment received any substantial support. For individual harvest treatments the mean growth rate was 4.25 cm/year for the adult harvest (n=119, SE=0.40), 3.46 cm/year for the control treatment (n=158, SE=0.32) and 2.94 cm/year for the juvenile harvest (n=116, SE=0.32). However, the differences in means between harvest treatments were an artefact of the different proportions of adults and juveniles remaining in quadrats and differences in growth rate disappeared, when the non-linear growth model was applied.

Fitted lines for each harvest treatment are shown in Figure 7. The 95% confidence intervals for estimates of parameters *a* (slope) and *b* (location of peak and asymptote along the x-axis) of the non-linear regression indicated no significant differences between treatments (Table 8).

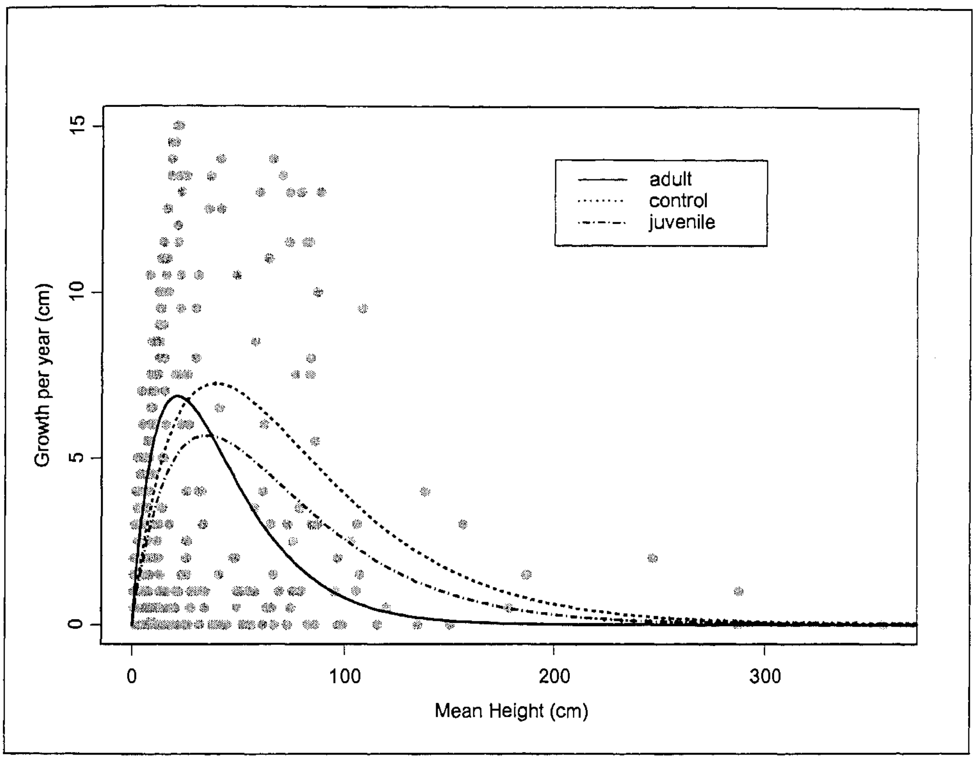

Figure 7. Fitted values for growth rates of *C. arnhemica* under three harvest treatments as a function of stem length using Hoerl's model. Values are corrected for site variation. The fitted lines are overlaid over observed growth rates for all treatments combined. Differences between treatments are not significant (see Table 8).

The preferred model, using the combination of fire and site effects, showed differences in growth between the three fire frequencies. Growth rates were highest in once burnt quadrats, followed by quadrats that had been burnt twice while unburnt quadrats showed the lowest growth rates (Figure 8).

Table 8. Parameter estimates and 95% confidence intervals for growth rates in different harvest treatments as determined by non-linear least squares regression using Hoerl's model for plant growth. Parameter *a* influences slope, parameter *b* influences location of peak and asymptote of the model. Residual SE = 3.64, df = 387.

Treatment	**95% confidence intervals for parameter estimates**	
	a	***b***
Adults	$0.53 \leq 0.87 \leq 1.21$	$15.15 \leq 21.38 \leq 27.61$
Control	$0.31 \leq 0.50 \leq 0.68$	$31.65 \leq 39.24 \leq 46.83$
Juveniles	$0.23 \leq 0.43 \leq 0.63$	$28.53 \leq 35.21 \leq 41.90$

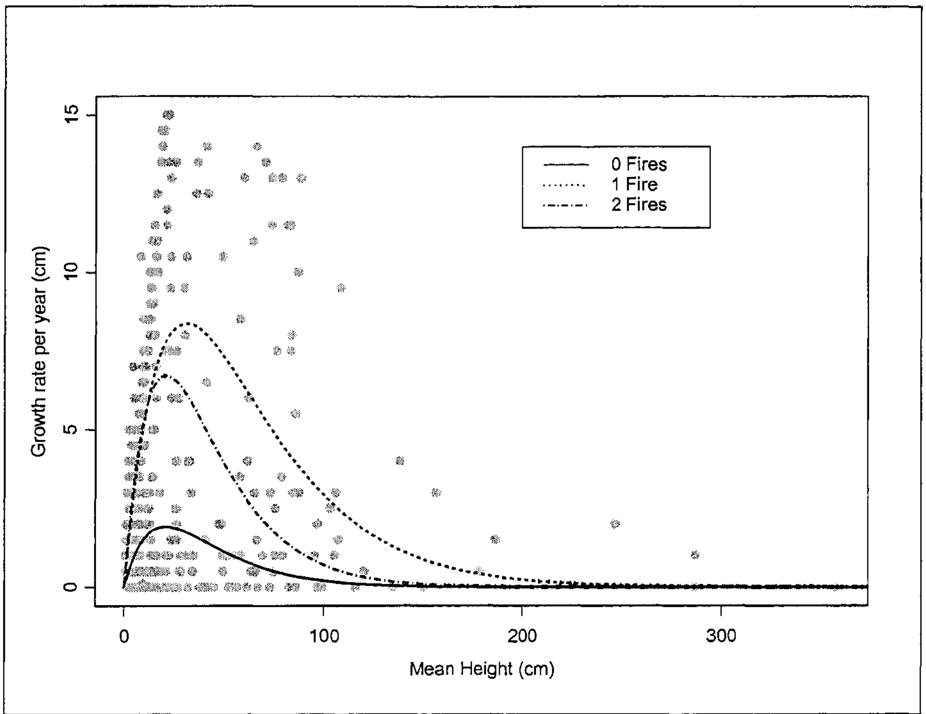

Figure 8. Fitted values for growth rates of *C. arnhemica* under three fire frequencies as a function of stem length using Hoerl's model. Values are corrected for site variation. The fitted lines are overlaid over observed growth rates for all fire frequencies combined. N(0 fires)=123, n(1 fire)=203,n(2 fires)=67.

There was a significant difference between growth rates in areas that had no fires compared to areas with two fires, as indicated by the confidence intervals of their parameter estimates for parameter *a*. The differences between areas with no fires and areas with one fire were marginally non-significant for parameter *a*. The estimates for parameter *b* were not significantly different for any of the different fire frequencies (Table 9).

Table 9. Parameter estimates and 95% confidence intervals for growth rates under different fire frequencies as determined by non-linear least squares regression using Hoerl's model for plant growth. Parameter *a* influences slope, parameter *b* influences location of peak and asymptote of the model. Estimates are corrected for site differences.

Fire	95% confidence intervals for parameter estimates	
	a	***b***
0	$-0.003 \leq 0.251 \leq 0.505$	$5.89 \leq 20.80 \leq 35.72$
1	$0.460 \leq 0.733 \leq 1.005$	$16.21 \leq 31.05 \leq 45.89$
2	$0.555 \leq 0.880 \leq 1.204$	$4.42 \leq 20.72 \leq 37.02$

3.4 Survival in Nursery

Nursery treatments had a marked effect on plant survival. The most parsimonious model to explain survival patterns of harvested *C. arnhemica* in the nursery was a combination of nursery treatment and plant size class. This model had a 54% likelihood (Akaike weight = 0.54) of being the most parsimonious model out of the candidate set. The next best survival model was the single-factor nursery treatment model with an Akaike weight of 0.44 (), followed by a combination of Treatment and Height (Akaike weight=0.02). All other models were highly unlikely. Table 10 shows a list of QAICcs, QAICc differences and Akaike weights for comparison of the full set of candidate models. According to Burnham and Anderson (1998), models outside the top 90% of Akaike weights have essentially no support. Therefore, only treatment and the combination of treatment and size class were considered further.

The single factor nursery treatment model was adopted as the preferred model because the reduction in deviance that was achieved by adding size class was negligible (Table 10). Plants that were stored bare-rooted for an extended period of time had a very low survival rate (0.18 ± 0.17) whereas survival rates were high in all other treatments including the control (0.73 – 0.83, SE: 0.17-0.20, Figure 9).

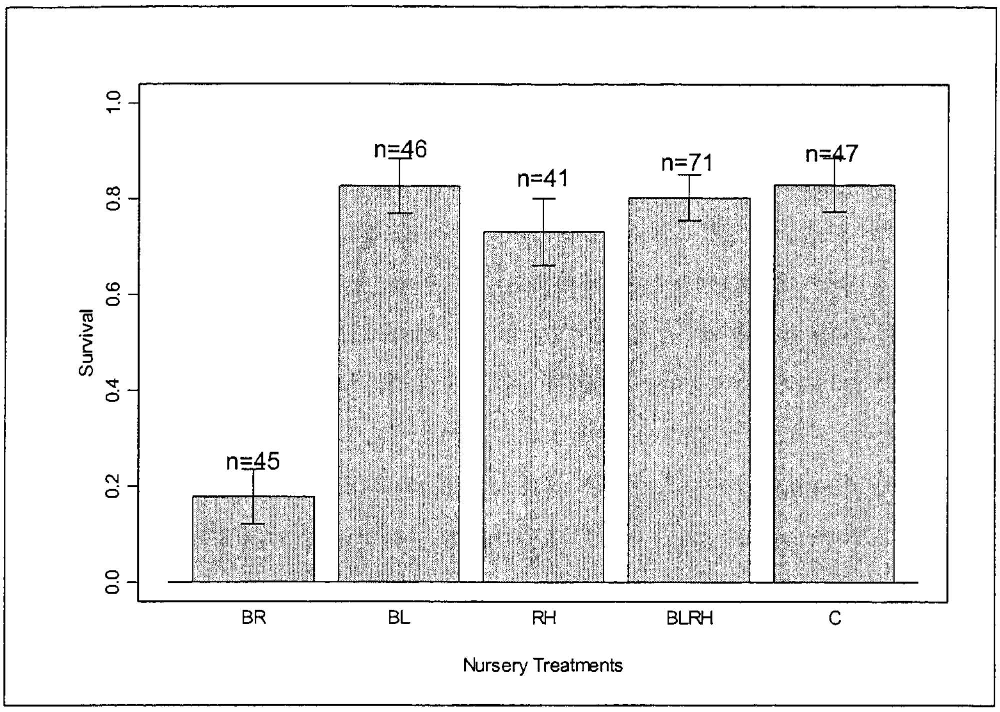

Figure 9: Survival under different nursery treatments over the two-year study period. Survival rate ± 1 SE. BR: Bare-rooted, BL: Bleach, RH: Rooting Hormone, BLRH: Bleach + Rooting Hormone, C: Control.

Table 10: Summary of candidate binomial generalised linear models for nursery survival. Model selection was based on the change in residual deviance from the null model (Null deviance = 310.34) using a chi-squared test, and quasi-likelihood modification of AIC (QAICc) to correct for overdispersion. Asterisks indicate significance of Chi-squared tests. Akaike weights in bold are models within the top 90% of weights.

Model	Factors	Residual df	Change in Deviance	p	ΔQAICc	Akaike weight
Null	-	249		-	45.19	0.0083 x10^{-8}
1	Nursery Treatment	245	65.64	<0.0001*	0.43	**0.44**
2	Size Class	248	0.01	0.92	44.73	0.010 x 10^{-8}
3	Site	246	6.07	0.10	43.45	0.019 x 10^{-8}
4	Harvest Date	248	3.80	0.05	43.19	0.022 x 10^{-8}
5	Height	248	4.97	0.02*	44.01	0.014 x 10^{-8}
6	Treatment + Size Class	244	64.71	<0.0001*	0	**0.54**
7	Treatment + Height	244	71.48	<0.0001*	6.49	0.02
Global	all factors	239	79.81	<0.0001*	18.42	0.029 x 10^{-8}

4 Discussion

4.1 Survival *in situ*

As long-lived plants cycads generally have high rates of survival (Jones 1993; Liddle 2003; Raimondo and Donaldson 2003; Whitelock 1975). *Cycas arnhemica* is no exception with high survival rates for juveniles and adults combined. Over the two-year study period overall survival was above 84.2 %, which is equivalent to an annual survival rate of more than 91.7%. These rates were similar to those found in a study of fire impact on *C. armstrongii,* a cycad found in similar habitat and of similar growth form to *C. arnhemica* (Liddle 2003). Under naturally occurring fuel loads, Liddle estimated survival rates from 81.4% for small juveniles and 96.6% for large adults.

The removal of individuals from a population through harvest may reduce intra-specific competition within the population and can lead to increased survival because resources are shared among a smaller number of plants or animals (Caughley and Sinclair 1994). There are currently no published studies on the effects of intra-specific competition on cycad life history although some studies have suggested that competition from neighbouring plants for light and nutrients may limit cycad growth (Clark and Clark 1987; Farrera*et al.* 2000). The harvest of juvenile or adult *C. arnhemica* did not influence the survival of the remaining individuals over the two years of this study. The number of plants removed for the experimental harvest was low in all treatments, as many plants were unsuitable for harvest. It is unclear if harvest effects would have become apparent if higher proportions of plants had been removed.

Fire seems to be more important as a determinant of cycad survival than harvest treatment in the short term. Fire is a major determinant of plant ecology in the northern Australian savannas and influences survival of many woody savanna species (Scholes and Archer 1997). Repeated burning has been associated with the decline of the Pacific cycad *C. seemannii* (Keppel 2002) and high fire intensities reduced both adult and juvenile cycad survival in *C. armstrongii* with the greatest reduction in survival recorded in the adult size classes (Liddle 2003). Out of the models considered in the present study, those that included effects of fire received considerably more support than the harvest model, suggesting that fire frequency is a more important factor in determining cycad survival than harvest regime. Many of the dead plants observed in our study were large adult cycads that had been severely burnt and of which only charred remains of the stem were found.

4.2 Recruitment

Recruitment of new seedlings into a population is influenced by the presence of reproductive individuals. Harvest reduces the number of reproductive plants in a population either immediately through the removal of mature individuals or after a time delay through removal of immature plants. Recruitment is variable in other cycad species (e.g. Watkinson and Powell 1997) and mast-seeding has also been observed in *C. armstrongii* (Liddle 2003).

Despite the removal of adult plants, no evidence was found that recruitment of new *Cycas arnhemica* seedlings was reduced in harvested areas. These results are less surprising considering that plants of more than 100 cm height were not harvested because it was not practical, so that many mature cycads remained in the quadrats. The smallest reproductive plant observed during the study was a male of 66 cm, all other reproductive plants were taller than 100 cm. The same trend was observed in *C. armstrongii* with the smallest reproductive plant being a 61 cm male and all others above 100 cm (Ornduff 1992). There might be a delayed effect on seedling recruitment that will only become apparent in the next generation of mature plants.

In the short term, other factors have a more pronounced effect on seedling establishment. In many woody savanna species seedling establishment is reduced where feral animal damage is high and the recruitment of seedlings may be inhibited by frequent fires (Setterfield 1997). Indeed, fire had a much stronger impact on seedling recruitment than harvest treatment in our study. Areas that were burnt twice during the study period contained no seedlings at all at the time of sampling in 2003 whereas once-burnt areas had the highest numbers of seedlings. The removal of grass and litter by fire may aid germination and initial seedling growth after the fire event. Reduced competition for light following fire has been implicated in the growth, reproduction and seedling recruitment of the cycads *Macrozamia riedlei* and *Zamia skinneri* (Clark and Clark 1987; Groveet *al.* 1980). This would explain the higher number of seedlings found in areas that were burnt once in the two years. Subsequent fires however, may lead to the death of new seedlings before they are able to establish underground stems from which to regenerate after a fire burns off leaves.

Over the longer term, seedling numbers alone do not give a complete picture of recruitment processes. Recruitment rate is not always related to the rate of increase of a population and seedling survival, establishment and transition to other life stages must be considered before any conclusions can be drawn (Caughley and Sinclair 1994).

4.3 Growth

Cycads are often described as slow-growing plants (e.g. Hill and Osborne 2001; Jones 1993; Thieret 1958) although this description is somewhat misleading and growth rates can differ widely between species (Whitelock 1975). The mean growth rate of 3.5 cm/year found for *C. arnhemica* in this study is similar to growth rates measured by Liddle (2003) and Watkinson and Powell (1997) for *C. armstrongii* of 3.5 cm/year and 4.5 cm/year respectively.

Similar to effects on survival, the reduction of competition for resources through harvest may lead to increased growth in many plant species (Crawley 1997). In this study however, harvest treatments had no impact on the growth of *C. arnhemica*. The mean growth rates for the three harvest treatments showed a decreasing growth trend from adult to control to juvenile harvest but these differences were an artefact of the different treatments. Where juveniles, which are faster growing, had been removed, the mean growth rate was lower, and likewise, where adults had been removed, the mean growth rate was higher. Using a non-linear growth function to model growth compensated for this effect and eliminated the differences.

Little is known about the factors that limit cycad growth in Australian savannas and limiting factors may differ between species, depending on habitat and growth form. There has been some suggestion that light is a limiting factor for certain rainforest cycad species (Clark and Clark 1988) and *Zamia pumila* (Negron-Ortiz and Gorchov 2000), although this seems unlikely for *C. arnhemica* which grows in open savanna woodland. Fire stimulates leaf growth in the first year after fire in the cycad *Macrozamia riedlei* (Groveet *al.* 1980). This is consistent with the pattern found for *C. arnhemica* which showed increased growth rates in quadrats that had been burnt.

4.4 Nursery Survival

The preferred model to explain differences in plant survival in the nursery did not include any of the environmental variables that were thought to potentially influence survival. Harvest date and site of origin did not influence plant survival. This has positive implications for harvesting practicalities because there is no disadvantage in opportunistic harvesting. The fact that the current sustainable use project is run as a community-based Aboriginal enterprise means that harvesting may not occur to a set schedule but is more likely to be carried out when and where an opportunity arises, for instance through the availability of a vehicle. Our results show no indication that this may be a disadvantage.

Results from the different nursery treatments showed that only one treatment, the bare-rooted storage of plants for an extended amount of time, affected plant survival. None of the three combinations of fungicide and rooting hormone had a significant impact on plant survival. This result is probably the more noteworthy in the context of feasibility of this community-based enterprise. Plants can be potted directly after harvesting without the need for costly and labour intensive treatments to increase plant viability. Costs of extraction and transport to markets are already high (Griffiths, unpublished data) so any reduction in production costs is important to keep the sale price of *C. arnhemica* low enough to attract consumer interest.

The bare-rooted treatment was applied to plants to assess the feasibility of exporting cycads overseas and interstate where soil cannot be transported with the plants due to export and quarantine regulations. Plants for export must be free of soil, seeds and other contaminants (Export Control Act 1982) and their root bulbs may therefore be exposed for the duration of transport. It is apparent that bare-rooted storage and transport, with less than 20% survivorship, is not a viable option and other methods will have to be explored if export is to be seriously considered.

4.5 Management implications

4.5.1 Harvest sustainability

It has recently been suggested that adult plant harvesting is detrimental to cycad populations (Raimondo and Donaldson 2003). The authors modelled the survival of two cycads species of differing life histories, *Encephalartos villosus* and *E. cycadifolius,* under a range of harvest intensities and estimated that populations would decline continuously if adult plants were harvested at an annual rate of 5%. They argue that, depending on the life history of the harvested cycads, only seed harvest or partial harvest of multi-stemmed cycads should be permitted to ensure the conservation of the species. Although this approach may be useful for endangered cycad species with small populations in which the harvest of a small number of plants represents a relatively high proportion of the population, the results cannot easily be applied to species with larger total populations like *C. arnhemica*. Raimondo and Donaldson's population model did not allow for a potential density-dependent increase in population growth after harvesting. Where models are based on life history data collected from harvested populations these often show a significantly greater harvesting tolerance than models based on data from unharvested populations (Freckleton*et al.* 2003; Ticktin*et al.* 2002).

Although no density-dependent effects were found in our study of *C. arnhemica*, it is still unlikely that harvest at the current level is detrimental for the species. Locally, the harvest rates in our study were high above those predicted to be fatal for *E. villosus* and *E. cycadifolius* populations, yet no impact could be detected on seedling recruitment or other life history traits. The results of our study suggest that the current rate of harvesting does not threaten the *C. arnhemica* population in Arnhem Land. Harvesting had no impact on *in situ* plant survival, recruitment or growth over the two-year study period and environmental effects, notably fire frequency, by far outweighed any effect of harvesting. Commercial harvest should therefore be allowed to continue.

Recommendation 1: The commercial harvest of *C. arnhemica* should be allowed to continue.

4.5.2 Harvest Rates

At the start of this study in 2001, the harvest limit for the commercial harvest of any Northern Territory cycad species was set to 500 individuals per year by the Parks and Wildlife Commission of the NT (PWCNT 1997). This number is very conservative and was probably chosen to ensure the sustainability of the harvest in light of a lack of scientific data on cycad population dynamics.

The Parks and Wildlife Commission amended its cycad management plan in September 2003 and revised maximum harvest limits upwards from 500 plants per year to 5% of the population. While 500 plants seemed a very conservative number for *C. arnhemica*, with a population that probably exceeds 5 million individuals (Griffiths, unpublished data), 5% of the population may well be too generous to ensure the survival of the species, considering the long time it takes for *C. arnhemica* and other cycads to reach maturity.

The maximum sustainable yield for a certain species depends on its rate of population growth (Caughley and Sinclair 1994). To reach a height of 100 cm *Cycas arnhemica* probably takes 25-30 years after emergence from the ground as indicated by the growth rates from this study. It is unknown how long it takes for the stem of the plant to emerge in the early stages of its life. Liddle (2003) also estimates that *C. armstrongii* may take at least several decades to mature. Although a comprehensive population model was not constructed for this project, it becomes clear that with the relatively low numbers of new recruits found for *C. arnhemica* and the long time to reach maturity, the population is unlikely to grow at a rate of 5% per year.

To ensure sustainability in the long term and to allow for stochastic events, such as droughts or cyclones, the precautionary principle dictates that the actual offtake should always be well below the maximum sustainable yield of a population (Caughley and Sinclair 1994).

The current maximum allowable harvest of 5% of the population is a theoretical upper limit and is subject to the discretion of the Director of the Parks and Wildlife Commission. Actual harvest limits are decided upon on a case by case basis for each application, so that it is unlikely harvesting at the maximum rate will ever occur. However, if the limit is set in the management plan, it is theoretically possible that future decision-makers might allow harvests up to this value and discretionary limitations are no substitute for a well-considered maximum harvest rate. Harvest limits in the management plan are meaningless unless they are set realistically.

The advantages and disadvantages of calculating sustainable harvest rates as a constant number or constant proportion must be taken into account when determining harvest quotas (Caughley and Sinclair 1994). If the constant number exceeds the maximum sustainable yield, the population will decline to extinction. This problem is less likely to be encountered with constant proportions, but they require the population size to be estimated annually to ensure calculations for the allowable offtake are correct. In the case of crocodiles, the Northern Territory government provides annual fixed quotas and undertakes monitoring of crocodile populations for the industry. However, resources to undertake such estimates are scarce and it is unlikely that these services would be provided for a commercial cycad harvest. A well considered constant number may be the more appropriate option for cycad harvesting, at least in the case of *C. arnhemica*, where the remoteness of the population makes accurate estimates of population size difficult and costly.

Recommendation 2: The current maximum allowable harvest of 5% of any cycad population, as set in the PWCNT cycad management plan, is too high and should be revised downwards.

4.5.3 Harvest of different life stages and repeated harvesting

The results from this and other studies suggest that it may be useful to impose restrictions on the harvest of certain life stages. This is already the case in other areas of management, including recreational barramundi fishing and the harvest of cycad seeds and leaves, where size limits and restrictions on repeated harvesting ensure that reproduction is not compromised (PWCNT 2003). Raimondo and Donaldson (2003) also suggest that the protection of adult plants is of paramount importance in the conservation of all cycad species. This suggestion seems justified in the case of *C. arnhemica* because recruitment of new seedlings into the population is slow. It does not conflict with the current practice of harvesting juvenile and small adult plants especially since this is the commercially more appropriate option. Seed dispersal appears to be very limited in *C. arnhemica* and most seedlings are found within approximately 1m of the mother plant (pers. obs.). This is consistent with observations by Watkinson and Powell (1997) who found seed dispersal in *C. armstrongii* to be restricted to less than 1 m, and (Burbidge and Whelan 1982) who investigated dispersal of seeds in *Macrozamia riedlei* which was less than 40 cm from the plant base. Therefore, when mature plants are removed, it is unlikely that recruitment will occur from elsewhere, even over relatively short distances.

Harvesting opportunities are limited to areas that are accessible with a vehicle and for practical reasons harvesting is likely to be concentrated close to vehicle tracks. There is some suggestion that localised over-harvesting of the rainforest carving wood species *Bombax ceiba* may occur in some patches close to Maningrida (Griffiths*et al.* 2003) although the limited occurrence of this species is very different to the widespread distribution and abundance of *C. arnhemica*. However, other studies have also indicated that harvesting intensities of high-value tree species may be higher in areas closer to population centres (Obiri*et al.* 2002).

Safeguards need to be included in the management plan, so that harvesting is spread over different areas and repeated harvesting in consecutive years is avoided. This will also limit the risk of over-harvesting of areas close to outstations and roads.

Recommendation 3: Repeated harvesting of the same areas and harvest of plants of more than 1 m stem height should be avoided.

4.5.4 Monitoring of cycad population

The annual cost of monitoring the population is high compared to the revenue that is created from the sale of *C. arnhemica* (Griffiths, unpublished data). However, effective scientifically-based monitoring is critical for the sustainability of any wildlife harvest and must not be driven by external factors (Bridgewater 1995). The cost is currently born by the Key Centre for Tropical Wildlife Management and the Bawinanga Aboriginal Corporation and partly funded from a research grant. In the future, the cost of monitoring the population will most likely be on the community. This option of private monitoring has been praised as a better alternative to state sponsored monitoring and works well elsewhere, for instance in the USA where goldenseal (*Hydrastis canadensis*) is harvested for medicinal purposes (Robbins 2000). Returns for these medicinal plants are large and a large industry is involved in the harvest unlike in the case of the Arnhem Land cycad. Unfortunately, it is unlikely that the Northern Territory government will take on the responsibility for monitoring *C. arnhemica* as it has done for commercial crocodile harvesting.

Monitoring efforts must be cost-effective to allow the enterprise to continue. The current study did not detect any adverse effects of harvesting on growth, survival or recruitment of *C. arnhemica* but it is important to continue monitoring of the cycad population to detect any long-term effects that may occur. Since the presence of reproductive individuals is essential for the continuation of the population, future monitoring efforts should be concentrated on this section of the cycad population. Survival of mature individual, their reproductive activity and recruitments into the mature size classes should be the focus of any future monitoring activity.

Recommendation 4: Survival and reproductive activity of cycads of > 1m stem length and recruitment into this size class ought to be monitored in harvested and unharvested areas.

4.6 Implications for the community

It is an essential feature of sustainable wildlife use that communities are an integral part of the decision-making process and derive an equitable share of the benefits from the exploitation of the wild species (Bridgewater 1995). The outcome of this study has important implications for the Aboriginal community that benefits from the harvest of *C. arnhemica*. The current harvest is not impacting on the population and the community can continue to derive income from the harvest without compromising conservation. Harvest rates can be increased if desired in accordance with the update management plan, so that revenue from the sale of more plants can be used for further research. The higher the commercial value of cycads for the community, the more likely they are to protect the plants. Although a land-use change that involves clearing of *C. arnhemica* habitat does not seem likely at present due to community values and perceptions, such options would lose some of their attraction in the future if income can be derived from the land and value is therefore added to the natural habitat of the cycad. Providing land-use alternatives is one of the basic features of sustainable use policies (Vardon 1995).

From the results of this study it appears unlikely that biological impacts on the cycad population will be a limiting factor for the cycad harvest venture. Other factors such as economic and cultural limitations may need to be examined to ensure the future success of the harvest enterprise. Efforts could now be concentrated on investigating market opportunities for the plants and their potential for export to overseas markets.

4.7 Limitations and further research

4.7.1 Limitations

Community involvement was a priority in this project and many different people were involved in the data collection. This compromised the accuracy of measurements and in some cases the assessment of whether a plant was alive or dead. Although measurement errors were probably consistent across all treatments, they led to high variability in the results. Survival estimates were also flawed by the fact that plants may have been counted as dead because they were either not visible due to the absence of leaves in small juveniles or their stems had been burnt and potentially live rootstocks had not had time to resprout.

The accuracy of the results for recruitment of new seedlings was compromised by the use of small sampling areas (20 x 5 m). Seedling distribution was very patchy and the numbers counted per quadrat were highly variable. Almost all new seedlings found during sampling were associated with a mature female plant and grew close to its stem. Mature female plants, however, were spread relatively widely and were often 5-15 m apart. Where no such plant fell into the sampling area, no seedlings were found. To obtain more accurate estimates of seedling numbers a different sampling regime must be applied, preferably using transects or quadrats of at least 20 x 10 m.

The experiment was not set up to investigate different rates of harvesting but rather the effects of harvesting different life stages. Harvest rates in adult and juvenile harvest treatments ranged from 10 to 32% and were limited by the suitability of plants for removal and commercial sale, so more intense harvesting was not investigated in this study. The results must therefore be considered to be valid only within the encountered range of harvest intensities.

The commercial harvest of *C. arnhemica* commenced in 2001 and data are limited to the two years that have passed since then. This is a relatively short period in the life of a long-lived species like *C. arnhemica* that probably takes several decades to mature.

4.7.2 Further Research

Longer-term studies are necessary to determine impacts of harvesting more accurately and to obtain more precise estimates of long-term survival and transition from juvenile stages to mature plants.

It is difficult to ascertain the true age of cycads and the time they take to reach maturity because of the prolonged period that seedlings spend growing an underground stem before an above-ground stem emerges and because of their ability to regenerate from seed stock. The time to emergence would have to be tested in field experiments to be able to predict cycad age more reliably.

The current method of monitoring the population is time-consuming and costly. It has been suggested that photographic monitoring may be an option, which would be cheaper, easier and faster. This option is currently investigated by the Key Centre for Tropical Wildlife Management but it is unlikely to be a substitute for more detailed population ecological studies. To make accurate predictions about the long-term survival of *C. arnhemica* populations it is necessary to investigate transition from smaller size classes into the larger ones more closely to form an objective picture of recruitment of plants into the reproductive life stages. The creation of such a transition model was beyond the scope of this project but may be an option for future research.

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