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An ecological risk assessment of the major weeds on the Magela Creek floodplain, Kakadu National Park

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1 Introduction

The negative impact of invasive weeds on ecosystem services and biodiversity is probably the most significant threat facing tropical wetlands today. Across the tropics there are many wetland weed species, some of them, including *Mimosa pigra*, *Salvinia molesta* and *Urochloa mutica* being widely distributed, if not pan-tropical. Such species have attracted a great deal of attention with the expenditure of large sums of money and effort on control techniques (Finlayson & Mitchell 1981, Storrs & Finlayson 1997, Douglas et al 1998). Fourteen of the top 18 environmental weeds in Australia invade wetlands (Humphries et al 1991). Twelve of these species are currently found in the Northern Territory. For Kakadu National Park, Storrs (1996) lists 15 species of high priority weeds that exist in small to large infestations and which are capable of significant impacts. KNP is thought to have up to 98 naturalised alien plant species (Brennan 1996); species which have become accepted and which have reproduced for several generations. This represents about 5.4% of the total flora but is relatively low when compared with an average of 21% in other Australian conservation areas (Lonsdale 1992a). The number of alien plant species in Kakadu has increased at the rate of 1.6 species per year since 1948, and is expected to continue as a result of increased tourism and development (Cowie & Werner 1993). For an area like Kakadu, apart from the problems of controlling existing weeds, there is the seemingly inexorable advance of major potential invaders (Storrs 1996).

The extent of invasion of wetlands by weeds has been described for some species although often incompletely. In many instances, vital information on the ecological changes wrought by these species is often confined to a few isolated studies or to anecdotal evidence. Economic analyses of the losses caused by pest species are not common. Additionally, studies on the social and cultural impacts of weeds have not been done (Finlayson & Spiers 1999).

A Global Biodiversity Forum held prior to the 1999 Ramsar Conference addressed invasive species and agreed upon the following definition: ‘An invasive species is a species, often alien, which colonises natural or semi-natural ecosystems, is an agent of change, and threatens native biological diversity’ (Pittock et al 1999). We accept this concept in our initial risk assessment with the additional recognition that invasive species also impact upon socio-economic values.

Given that weeds are an increasingly serious problem in tropical wetlands, there is a need for management prescriptions to be developed at several levels. Critically, for managers and users of wetlands, practical techniques and options are required that take into account local differences, priorities and resource levels for control. However, for localised effort to be effective a strategic framework is required that provides the necessary options and places particular weed infestations and their control into a regional perspective. A means of ensuring that the above aspects are not forgotten is through the adoption of ecological or wetland risk assessment procedures as the basis for effective and strategic weed management.

Within this context, information on the biology, ecology and management of *Mimosa pigra* (mimosa), *Salvinia molesta* (salvinia) and *Urochloa mutica* (para grass) has been collated and analysed in a risk assessment of the weeds in the regional context of Kakadu National Park, with particular emphasis on the Magela Creek catchment.

1.1 Preface

This report is an interim progress report only. The final risk assessment will contain additional information as outlined in section 6. Much of this information is available in references already obtained for the project, but is yet to be placed in a risk assessment context. Further modelling will be done when additional data on the distribution of weed species, plot sizes and details of survey and control effort, and costs, are obtained.

1.2 Background

In October 1998 the World Heritage Committee (WHC) mission to the Kakadu National Park (KNP) World Heritage site expressed concern over the possible impacts on natural and cultural values resulting from a proposal to mine uranium at Jabiluka (The Jabiluka Mill Alternative). At the request of the WHC, the Australian Supervising Scientist (SS) reported to the committee on the scientific concerns raised. The WHC obtained the support of the International Council of Science Union (ICSU) to form an Independent Science Panel (ISP) to review this SS report (ISP 2000).

The ISP concluded that the SS report had reduced the scientific uncertainties but that issues remained which needed additional analysis and/or clarification. The ISP made 17 principal recommendations but noted that its insights had been limited by lack of time and the need for both a site visit and further information. In July 1999 the WHC requested the ICSU to continue the work of the ISP in co-operation with the SS and the World Conservation Union (IUCN) in an attempt to resolve the remaining scientific issues (ISP 2000).

The ISP and (IUCN) stated that, whilst impacts from the site-specific Jabiluka mine proposal were most likely very small or negligible, a more comprehensive risk assessment of both the freshwater and terrestrial ecosystem at a *landscape-catchment* scale was needed. This was because the Alligator Rivers Region (ARR) is subject to major seasonal or long-term changes unrelated to those which may arise from mining impacts. For example, the impact of invasive weeds is probably the most significant non-mining threat facing the Magela Creek wetland system at the landscape-scale. Hence, they suggest that comprehensive monitoring programmes and accompanying analyses (assessments) are needed to distinguish between impacts from these differing causes and any unforeseen problems arising from. The ISP recommended also that research be run for several years before mining starts. The review panel clearly invites Environment Australia - Environmental Research Institute of the Supervising Scientist (***eriss***) and Parks Australia North (PAN) to undertake inventory, assessment and monitoring activities at landscape scales in order to guide future ecosystem management.

1.3 Project aims

The three major weeds (mimosa, salvinia and para grass) that occur on the floodplains of the Magela Creek system will be assessed using the wetlands risk assessment (WRA) framework proposed for wetlands by van Dam et al (1999). The WRA will address four main questions:

1. what areas of the Magela Creek floodplains (macro-habitats) of KNP are at risk of invasion by each of the three weed species? (i.e. current distribution & trend analysis where feasible);
2. what are the likely consequences of these invasions? (i.e. assessment of likely effects & pressures);
3. what management actions are being undertaken, or need to be undertaken to minimise the risks of further invasions across the Park and region (ARR)?; and
4. determine how the presence and spread of these species could confound assessment of any mining-related pressures.

1.4 Approach

1.4.1 Wetland risk assessment framework

Over the last decade the concept of environmental risk assessment has developed and expanded from a narrow and precise analysis of quantitative ecotoxicological data to more general and qualitative/semi-quantitative analyses of environmental problems (van Dam et al 1999). This has led to the Ramsar Convention on Wetlands recommending a model for wetland risk assessment (Figure 1), coupled with advice on the deployment of early warning systems for detecting adverse ecological change in wetlands. The Ramsar procedures are linked with a concurrent effort to espouse the values of wetlands and the maintenance of their ecological character. The former have been summarised by Finlayson (1996), as outlined below.

- *Functions* performed by wetlands are the result of the interactions between the biological, chemical and physical components of a wetland, such as soils, water, plants and animals, and include: water storage; storm protection and flood mitigation; shoreline stabilisation and erosion control; groundwater recharge; groundwater discharge; retention of nutrients, sediments and pollutants; and stabilisation of local climatic conditions, particularly rainfall and temperature.
- *Products* are generated by the interactions between the biological, chemical and physical components of a wetland, and include: wildlife resources; fisheries; forest resources; forage resources; agricultural resources; and water supply.
- *Attributes* of a wetland have value either because they induce certain uses or because they are valued themselves, and include the following: biological diversity; geomorphic features; and unique cultural and heritage features.

The combination of wetland *functions*, *products* and *attributes* give the wetland *benefits and values* that make it important to society.

In the context of the Ramsar Wetlands Convention and the wise use of wetlands it is stressed that the use and management of a wetland and its resources should be done in a manner that is consistent with the maintenance of the ecological character of the wetland. Ecological character is now defined by Finlayson (1996) as:

> the sum of the biological, physical, and chemical components of the wetland ecosystem, and their interactions which maintain the wetlands and its products, functions and attributes.

The generic wetland risk assessment model recommended for the Ramsar Convention has been derived from those used for water pollution and ecotoxicological assessments (eg USEPA 1998) as well as the more general methods developed for assessing the vulnerability of wetlands to climate change and sea level rise. The model provides guidance for environmental managers and researchers to collate and assess relevant information, and to use this as a basis for management decisions that will not result in adverse change to the ecological character of the wetland. Our objective has been to provide a framework for informed decision-making. Thus, it is not prescriptive. It is important to note that this assessment addresses the first four steps of the WRA process. The risk management process, which is a separate undertaking, is the responsibility of the relevant agencies.

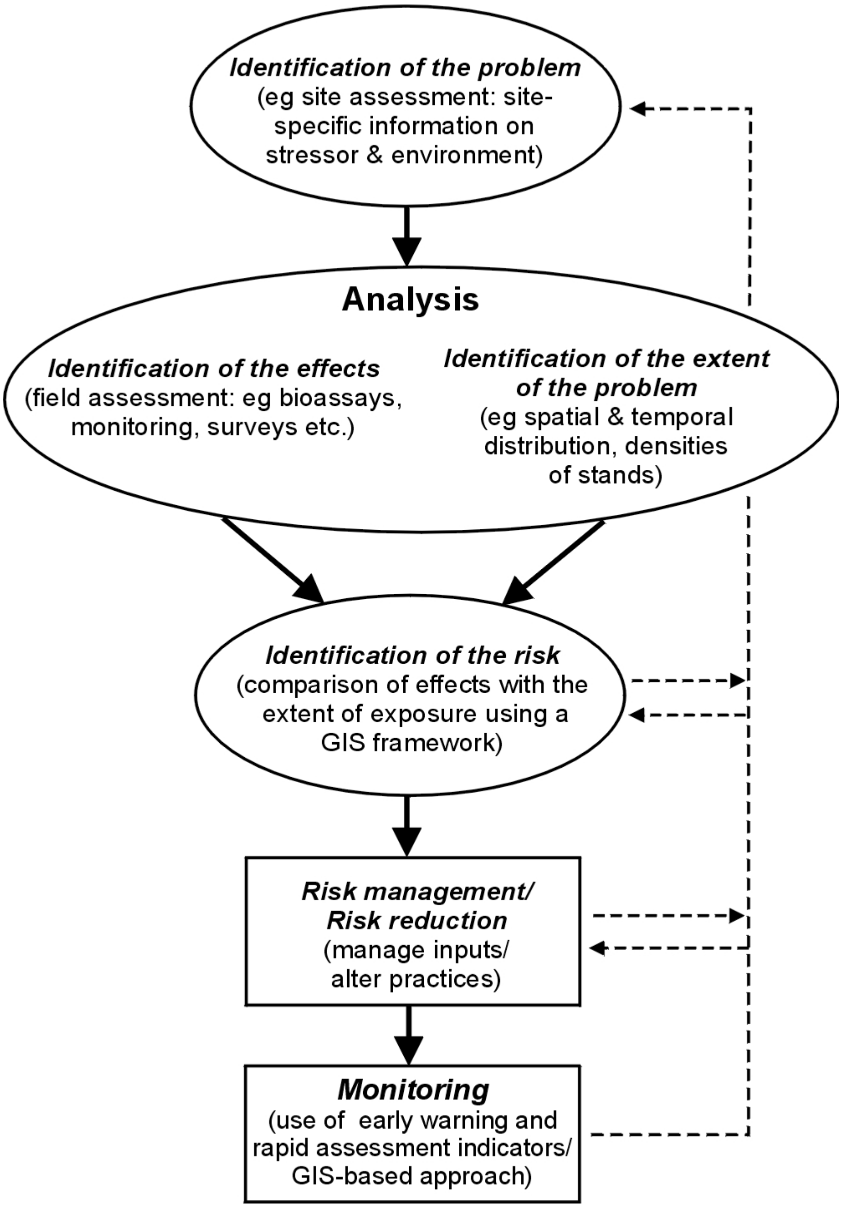


Figure 1 Wetland risk assessment framework (adapted from van Dam et al 1999)

The major steps in the WRA process presented in figure 1 are briefly described below, and are adapted from the Ramsar Convention Bureau (2000) and van Dam et al (1999).

1. ***Identification of the problem:*** what is the nature of the problem and how do we plan for the remainder of the risk assessment based on this information. Define the objectives and scope of the risk assessment.

2. ***Identification of the effects:*** evaluate the likely extent of adverse change or impact on the wetland, including cultural values in consultation with Traditional Owners (e.g. sacred sites). Where possible field studies will be used and are preferable for assessments of multiple impacts (as occurs on many wetlands). However, literature reviews of existing information may often be sufficient to identify some impacts.

3. ***Identification of the extent of the problem:*** estimate the likely extent of the problem on the wetland(s) of concern by using information gathered about its characteristics and extent of occurrence elsewhere. Map current distribution in order to estimate its potential distribution (& hence threat).

4. ***Identification of the risk:*** integrate results from the assessment of the likely effects with those from the assessment of the likely extent of the problem, in order to estimate the likely level of adverse ecological (& social) change within the Magela floodplain. A GIS-based approach can be a useful technique for characterising risks to wetlands (see below), by overlaying relevant information onto a map of the region of interest in order to link effects to extent/distribution. In addition to estimating risks, focus future assessments and/or monitoring on identified problem areas. The uncertainty and information gaps associated with the assessment will be described.

5. ***Risk management and reduction:*** the final decision-making process uses information obtained from the assessment processes described above, and in conjunction with other relevant information (eg political, social, economic, and engineering), and attempts to minimise the risks without compromising other societal, community or environmental values. It is a multidisciplinary task usually requiring coordination by resource managers and communication between stakeholders.

6. ***Monitoring & trend analysis:*** This is the last step in the overall risk assessment process and will be undertaken to verify the effectiveness of the risk management decisions. It will incorporate components that function as a reliable early warning system, detecting the failure or poor performance of risk management decisions prior to serious environmental harm occurring. The risk assessment will be of little value if effective monitoring is not undertaken. Trend analysis will likely require 4 or 5 years of further data on distribution and abundance of three major weeds, and their annual rates of spread (using mapped vegetation types & spot GPS locations from historical data and/or recent PAN weed surveys).

1.4.2 Information sources

1.4.2.1 Literature review

Published and unpublished reports were sourced and obtained through a comprehensive literature review process. Further relevant publications were then identified and obtained from within these sources.

1.4.2.2 Data sources and status of data

Weed data for Kakadu National Park were obtained from all of the district databases except for Jim Jim, where no data were available. Most data were in Fugawi® software as waypoint files and some additional data were available as hard copies. All data were converted or transcribed to spreadsheets and imported into the ArcView™ desktop Geographic Information System (GIS). The KNP mimosa database, which spans 1981 to the present, was also obtained and relevant information was extracted and entered into the GIS. Information from this database was also collated and used for preliminary cost of control modelling (see section 5.1.1).

The Park has a new weeds database that will centralise all existing and future data for the entire Park, and which will form the basis of a GIS. There were no records in this new database at the time of data acquisition for this project.

Additional data were obtained from the Northern Territory Department of Infrastructure, Planning and Environment (DIPE), and Energy Resources of Australia (ERA) supplied mine-site weed data for Ranger and Jabiluka.

eriss holds historical Magela floodplain para grass data, collected for the purposes of an ***eriss*** research project (Knerr 1998). Fieldwork conducted by ***eriss*** in March 2003 yielded some updated information on the distribution and abundance of para grass and salvinia.

1.4.2.3 Map information

The current weed data is presented on an IKONOS® image flown in June 2001. It is multispectral and sharpened with 1m pan band using the 'Hue, Saturation, Value' (HSV) technique. As IKONOS images require large amounts of disk space, a quick look JPEG file was used which has degraded image quality (Pfitzner 2003). It is anticipated that new aerial photographs at a scale of 1:25 000 will be available early in 2004, and the data in the final report will be presented on these new images.

2 Identification of the problem

2.1 Mimosa

2.1.1 Physical description

The following description is summarised from Lonsdale (1992b) and Miller et al (1981).

When mature, mimosa is an erect much branched prickly leguminous shrub reaching a height of up to 6 m, reproducing by seed and suckers. Mature plants branch from the base, and in seasonally inundated areas a skirt of fibrous adventitious roots is formed. The major root system consists of a 1–2 m taproot and lateral roots that extend up to 3.5 m from the stem. Stems are up to 3 m long, greenish at first but become woody, with randomly scattered slightly recurved broad-based prickles 5–10 mm long. The leaves are bright green, 20–25 cm long and bipinnate consisting of about 15 pairs of opposite primary segments about 5 cm long. Each segment has 20–42 pairs of sessile, narrowly lanceolate leaflets (3–8 mm long) per pinna that fold together when touched or injured and at night. Pairs of prickles sometimes occur between the branchlets on the main leaf stalk.

The flowers are pink or mauve, small, regular and grouped into globular heads 1–2 cm in diameter, each head containing approximately 100 flowers. The heads are borne on stalks 2–3 cm long, with 2 in each leaf axil, while the corolla has 4 lobes with 8 pink stamens. The fruit is a thick hairy, about 20–25 seeded flattened pod borne in groups (about 7) in the leaf axils, each 3–8 cm long and 7–14 mm wide. The fruit turns brown when mature, breaking into one-seeded segments. The ripe seeds are brown or olive green, oblong, flattened, 4–6 mm long, about 2 mm wide and weigh between 0.006 and 0.17g. The whole process from flower bud to ripe seed takes about 5 weeks.

2.1.2 Biology

Germination of the hard seeds is dependent on breaking down the physical barrier to moisture formed by the impermeable seed coat, with soil abrasion, microbial action, temperature fluctuations (Lonsdale et al 1988, Lonsdale 1993b) and, in some cases, fire most likely to breaks seed dormancy (Miller & Lonsdale 1992). Seed viability in excess of 5 years has been observed under laboratory conditions, and under natural conditions, seed half-life varies from 9 to 99 weeks, depending on the soil type and depth of burial. Seed production has been measured at between ~9000 and ~11 000 m^{-2} per year (Lonsdale 1988, Lonsdale et al 1988).

Plants can survive the dry season by steadily losing leaves, while in permanently moist sites, growth and flowering can continue more or less all year round (Wanichanantakul & Chinawong 1979). In the more dense stands, plant densities are in the order of 1–3 m^{-2} (Lonsdale et al 1995). There is a strong seasonality in growth rates with maximum rates in the field measured at 1.33 cm d^{-1} for seedlings, and 1.1 cm d^{-1} for plants >12 months old (Miller 1988). The main growth period is in the wet season with new shoots appearing with the first rains and a dense canopy forming within about a month.

Under ideal conditions, plants can begin flowering 6–8 months after germination. The main flowering period is the mid to late wet season, but flower production may continue as long as water is available (Lonsdale 1988). Flower bud maturation usually takes 7–9 days from bud formation. Mature seedpods develop 25 or more days after the flower buds mature, with peak seed falls occurring between the late Wet and early Dry seasons (Lonsdale 1988).

2.1.3 Advantageous features

Mimosa has many features that are generally considered ‘advantageous’ to a weed. The greatest problem for plants growing in flooded soils is that their roots drown in the anaerobic conditions. Mimosa withstands such conditions by sprouting adventitious roots near the surface where they can take up oxygenated water (Miller et al 1981). Thus, the thickets can continue their advance until only a tiny remnant of open water remains in the deepest parts of the billabongs (Braithwaite et al 1989). Mimosa also has the potential to invade tidal zones (Miller 1983).

The plants mature quickly and can set seed in their first year of growth (Lonsdale et al 1985). The seeds of mimosa are well designed for easy and rapid dispersal. The seedpods break into segments when mature, with each segment containing a single seed. These segments are covered with bristles that enable them to adhere to animals and clothing, and to float on water for extended periods (Miller et al 1981, Lonsdale 1993a). The seeds are also dispersed in soil and mud, adhering to vehicles, machinery (Lonsdale et al 1985) and boats. Livestock and native animals sometimes graze mimosa plants (Miller 1988) and pass the seeds in their dung (Miller & Lonsdale 1987). Although spending up to 36 hours in the gut of an animal, 70–90% of mimosa seeds may still be viable (Benyasut & Pitt 1992).

The lifespan of the seeds in the ground depends greatly on their depth in the soil and the soil type. For example, half of a seed population was no longer viable after 99 weeks at a depth of 10 cm in a light clay soil, while a similar loss in viability was observed after only 9 weeks in a heavier cracking clay (Lonsdale et al 1988). In sandy soils, observations suggest that seed lifespan may be as high as 23 years (Lonsdale 1992b).

Regular heating and cooling of the soil surface results in a soil temperature range from about 25 to 70°C, causing expansion and contraction of the hard seed-coats of mimosa species, eventually making them crack, breaking their dormancy. The deeper in the soil a seed lies, the less extreme is the temperature range. Thus, seeds buried deeper than 10 cm cannot successfully germinate (Lonsdale 1993b). However, as they can remain viable for long periods, such seeds could eventually germinate if brought to the surface by cultivation or disturbance by animals, even if all the adult plants are removed (Lonsdale et al 1988).

The rate of seed production has been measured between ~9000 and ~11 000 m^{-2} per year depending on environmental conditions (Lonsdale et al 1988). The most productive plant observed in the field in Australia had a crown of about 8 m^2 and produced about 11 000 pods per year, equivalent to about 220 000 seeds (Lonsdale 1992b).

The compound leaves of mimosa, like those of several other species in the genus, close in response to electrical, mechanical, thermal and light stimuli and wounding (Simons 1981). This may protect the leaves from damage in certain circumstances. This feature has also greatly assisted in the spread of the weed as humans value this novelty aspect and transported mimosa vast distances as a garden ornamental.

Under the right conditions mimosa grows quickly at a rate of about 1 cm per day, and infestations can double in area in one year. It can also withstand droughts, so the extended dry season, although slowing the growth rate and thinning the canopy, does not kill mimosa (Lonsdale 1993a).

If chopped down mimosa will easily resprout from the stump (Wanichanantakul & Chinawong 1979). If mimosa is burnt, the foliage may become desiccated and fall, but up to 90% of mature plants and up to 50% of seedlings may regrow, probably from dormant buds at the base of the plant (Miller & Lonsdale 1992).

Mimosa has low nutrient requirements and consequently can grow within a wide range of soil types including nutrient poor sands, alluvial red and yellow earths, silty loams and heavy black cracking clays (Miller 1983).

2.1.4 History of mimosa invasion

Mimosa was first discovered in KNP in 1981 (Skeat et al 1987) and by 1993 a total of 160 outbreaks were recorded in the Park (Cook et al 1996). The suspected seed source for the outbreaks in central KNP could have been either of the two large infestations to the east and west of the Park. The Oenpelli stand to the east was doubling in area every 1.4 years resulting in an increase from 200 – 6000 ha between 1984 and 1991. In 1991 dense mimosa covered approximately 40 km^2 on the Mary river floodplain (Cook et al 1996). Many of the outbreaks in KNP could be attributed to movement of large mammals including feral pigs and buffalo, with subsequent dispersal by water (Lonsdale & Lane 1994). Waterbirds may have been responsible for dispersal of seeds to some of the more remote sites (Cook et al 1996). Feral water buffalo were not reduced in large numbers from the Park until 1990 (ANPWS 1991), so prior to this there was a high level of ground disturbance that facilitated the establishment and spread of mimosa. The highest densities of mimosa outbreaks in KNP were found in the wetland margins that had been disturbed by feral animals, and in wetland habitats with a relatively long period of inundation, but where trees were absent (Cook et al 1996). The systematic survey of the Park and the destruction of new outbreaks have prevented the establishment of mimosa within the Park. Seedlings have emerged for up to 10 years after treatment of existing sites (Cook et al 1996), so long-term and ongoing follow-up treatment continues.

2.2 Para grass

2.2.1 Physical description

(adapted from Cowie et al 2000)

A trailing stoloniferous perennial grass 0.6 – 3m tall. The culms (stems) are hollow and robust, creeping in a prostrate growth habit. These stems stand erect towards the ends, and sprout new roots wherever the nodes touch the ground. Leaf blades are hairy and dark green in colour and may be 6 – 30 cm long and 0.5 – 2 cm wide, tapering to a long, fine point. The leaf sheaths are also hairy, particularly where they join the stem. Florets (flower heads) are approximately 18 cm long with up to 10 racemes, each about 3 – 9 cm long.

2.2.2 Biology

Para grass thrives under conditions of high temperature, high humidity and elevated soil moisture (Whitehead 1992). Growth is best on sites that undergo relatively shallow inundation (Calder 1982, Wildin 1991). It also grows well on floating vegetation mats (Wilson et al 1991). It was thought previously that para grass did not grow in water depths greater than 50–60 cm (Anning & Hyde 1987). However, studies by Douglas et al (2001) frequently recorded para grass growing in depths greater than 2 m on the Magela floodplain. Flowering and fruiting is from April to November, although reproduction is predominantly vegetative, with relatively little seed produced (Cowie et al 2000, Wesley-Smith 1973).

There is little published information on seed production and viability of para grass. Knerr (1998) found that on the Magela floodplain, floret production was in the order of 12 000 m^2 during the peak biomass period (May), and there were an estimated 7000–8000 seeds per m^2 in the sediment seed bank. Less than 3% of para grass seeds germinated after three weeks, however, about 80 – 90 % of the remaining ungerminated seeds were interpreted as viable (using tetrazolium chloride). This indicated that the seeds have a dormancy mechanism, which has been suggested as an impermeable seed coat (MacLean and Grof 1968). The maximum lifespan of the seeds in the seedbank is unknown.

2.2.3 Advantageous features

Para grass is capable of withstanding prolonged immersion at a range of water depths up to >2 m. It can occupy a broad range of habitats from the *Melaleuca* woodland and swamp forests that fringe the floodplain, to the edges of permanent floodplain waterbodies (Knerr 1998, Douglas et al 2001). Para grass is also tolerant of drought and of brackish water, but is susceptible to frost (Holm et al 1977). It's ability to invade the floating vegetation mats found in many Top End billabongs means that vegetative reproduction can occur over greater distances as these mats often break away and float to new areas (Hill & Webb 1982, Hill et al 1987).

Para grass is able to rapidly recover from disturbance such as grazing by animals (Clarkson 1991), and has been observed to re-shoot vigorously following fire (Knerr 1998, Douglas et al 2001). Its greater biomass can change fuel dynamics in such a way as to facilitate its spread. Native grasses may be susceptible to hot fires (Whitehead and McGuffog 1997) and the spread of para grass may be aided by its ability to recover rapidly from hot fires and thus invade other grassy habitats (Douglas et al 2001).

In the Wet season, para grass has a greater leaf to stem surface area ratio compared with other native floodplain grasses that occupy a similar niche. The greater proportion of leaf area gives para grass the potential to more efficiently convert solar energy to biomass, possibly increasing it's invasive potential (Douglas et al 2001).

The value of para grass as an animal fodder species has greatly assisted its spread. Humans have moved it over great distances to areas where, under normal circumstances of spread, it would never have invaded. This has the attendant risk of carrying other aquatic weeds, pathogens or animal pests between catchments (Clarkson 1995).

2.2.4 History of para grass invasion

Prior to the declaration of Kakadu National Park in 1979, pastoralists and Government departments planted para grass as a fodder species in several areas in the Alligator Rivers Region (Wesley-Smith 1973, Miller 1970). The first introduction was in 1922 when a couple of acres were planted at Mission Billabong near Oenpelli. By 1960 an estimated 3100 ha of the Oenpelli floodplains was infested (Letts 1960).

It was first reported in the Magela catchment in 1946 on the Cannon Hill floodplain (Christian & Aldrick 1977) where it now covers a large area of the surrounding floodplains. On the Magela floodplain it was first observed in the 1950's. In 1968/69 the NT Government trialled a number of pasture grasses at Mudginberri, with para grass being one of these species (Miller

1970). It is now widespread on the floodplain (~920 ha in 1996) with various sized infestations found from Mudginberri to the north of Nankeen billabong where the largest infestation exists (Knerr 1998, Salau 1995). It has also been reported on the East Alligator bank at the Magela Creek outflow (Salau 1995) .

In 1961, a pastoralist collected para grass runners from the Botanical Gardens in Darwin and planted them around the Munmarlary area in the South Alligator River catchment. The same NT Government department that conducted the trials at Mudginberri in 1968/69, also trialled the grasses at Munmarlary (Salau 1995). Para grass has since expanded from this small area to cover a larger areas of the South Alligator floodplains.

To the west, a few para grass runners were planted at the bottom crossing of 4-Mile hole. The infestation has expanded over the years and satellite infestations are scattered for approximately two kilometres down the floodplain (Salau 1995).

In the Nourlangie area, para grass was planted at No. 2 Goose Camp sometime in the 1970's. Treatment in the early 1990's successfully controlled this infestation (Salau 1995).

Other infestations around the Park have been located in Diddy Gee Gee swamp (Magela catchment), Boggy Plain (South Alligator catchment), Jono's Jungle (East Alligator catchment) and in Stage 3 in the South Alligator River bed upstream from the Sleisbeck road crossing. Most of these infestations have been successfully treated and are being monitored (Salau 1995).

2.3 Salvinia

2.3.1 Physical description

(adapted from Cowie et al 2000, Room & Julien 1995, and Storrs & Julien 1996)

Salvinia is a free-floating perennial aquatic fern with a horizontal rhizome that lies just below the surface of the water. A pair of floating leaves is produced at each node and a submerged, highly divided, modified leaf that resembles and functions as a root, but there are no true roots. Individual plants are usually less than 30 cm long. The floating leaves are green in colour, and ovate to oblong in shape. They are covered in rows of waxy bristly hairs, the stalks of each dividing into four thin branches that rejoin at the tips to form a cage. The resulting structures resemble tiny eggbeaters. These specialised hair structures provide a water repellent protective covering. The lower surfaces of leaves are covered with fine, sharply pointed hairs, which are darker in colour. These are particularly dense on the submerged leaves. Racemes of papillate sporocarps are borne among the filaments of the 'roots'.

There is considerable morphological variation, with crowding and nutrient availability the most important determining factors, their effects being largely dependent of one another. Three growth forms (with a continuum between them) have been described where individual leaves can range in size from a few millimetres to up to 6 cm in diameter. During early colonisation, small, thin oval leafed (about 1 cm diameter) plants lie flat on the water surface. As populations expand the leaves curl slightly at the edges in response to self-competition. At this stage the leaves may be about 2 cm in diameter and the entire lower leaf surface is in contact with the water. The tertiary or mat-form is produced when the plants are crowded in mature infestations. The leaves are acutely folded along the mid-rib giving a vertical appearance. They are heart shaped or oblong and up to 6 cm in diameter.

2.3.2 Biology

(adapted from Cowie et al 2000, Room & Julien 1995, and Storrs & Julien 1996)

Salvinia is sterile and reproduces vegetatively. Fragmentation occurs when the main rhizome breaks at the nodes as plants mature, and new branches develop from the apical and lateral buds. Each node can harbour up to five serial lateral buds. Damaged or dead plant material stimulates the development of the buds into branches, and the process continues.

Specimens of salvinia from different habitats and countries appear to be genetically identical (ie no somatic mutations have been found). Thus the entire species may be a single clonal genet (single genetic individual) and possibly the largest single (fragmented) organism on the planet.

Under ideal conditions and away from natural enemies, salvinia is capable of doubling its dry weight every 2–3 days (Finlayson 1984b). For KNP, the fastest recorded growth rate was a doubling of dry weight every 5–7 days.

Growth is best in still or slow-flowing fresh waters of the tropics and sub tropics. Although it prefers more eutrophic water where nutrients are not limiting, it will grow in clear water and in waters subject to flooding. In faster flowing waters, salvinia exists as an understorey to other vegetation that holds it in place.

The best growth rates are achieved at a temperature range of about 20–30°C, with the optimum being 30°C. Little growth occurs below 20°C and laboratory experiments (2 hours at -3°C) killed all buds. Growth is thought not to occur over 40°C, and all buds die when exposed for 2 hours at 43°C.

Growth rates of salvinia are significantly affected by nutrient availability, particularly nitrogen and phosphorus. High nitrogen levels in particular increase both the rate of extension of existing branches and the rate of production of new branches.

2.3.3 Advantageous features

(adapted from Cowie et al 2000, Room & Julien 1995, and Storrs & Julien 1996)

Salvinia exhibits a remarkable ability to regenerate after being severely damaged or desiccated. Green shoots may appear on plants that appear completely dead. For example, it can survive for several months on floodplains that dry out annually. Exposed plants on the surface may desiccate, but plants underneath or on the moist ground can survive. A similar scenario applies to occasional exposure to freezing temperatures where exposed floating leaves may die but leaves lower down in the mat survive, and the buds can remain viable as long as they are protected.

Comparisons between nutrient requirements in the laboratory and nutrient concentrations measured in the field, demonstrate that salvinia is almost always growing under conditions of limited nutrients. To offset this, the plant has become adapted to low ambient nutrient levels. It can mobilise nutrients from senescent tissues for use by growing points and can take up nutrients from ephemeral flushes carried in runoff from rainfall, storing excess for future growth.

Following separation from the parent plant, a number of branches can develop at each node and give rise to individual plants. Salvinia's ability to grow from the smallest of fragments and its rapid growth rate (see section 2.2.2) make it an aggressive and competitive species.

Like mimosa, salvinia is a species valued by humans. Its attraction as an aquarium and pond ornamental has resulted in large-scale movements. Once relocated, salvinia may enter the wild via deliberate or inadvertent releases.

2.3.4 History of salvinia invasion

Salvinia was first recorded in the Park on the 5th September 1983 in several billabongs and interconnecting channels downstream of the Oenpelli road crossing of Magela Creek, a tributary of the East Alligator River (Finlayson 1984a). The infestation covered approximately 7 km^2, and observations on the distribution and presence of dead plants in the trees above the water line indicated that the weed was present during the previous wet season (Storrs & Julien 1996). A small infestation of water hyacinth (*Eichhornia crassipes*) was discovered close to the road at the same time, and fortunately this was removed and apparently eradicated (Finlayson 1984a).

The location and the presence of the two weeds together gave rise to speculation that the weed was introduced by humans (Finlayson 1984a), either deliberately or possibly by viable plant material being washed from a vehicle or boat. This was the third time in two years that that field infestations of the two weeds had been found together in the NT (Storrs & Julien 1996). Early attempts to remove salvinia failed and over 2–3 years it spread throughout the Magela floodplain and all the main billabongs were at different times at least partly, or even completely covered by floating mats of plants (Finlayson et al 1994). By the late 1980’s salvinia had spread upstream of the Magela Creek crossing into Mudginberri Billabong and surrounding swamplands, and in 1990 the whole of the Magela floodplain was quarantined (Storrs & Julien 1996).

Despite quarantine efforts a new infestation was discovered in the Mekinj Valley in Tin Camp Creek, another tributary of the East Alligator River. This infestation also escaped during ensuing wet seasons and spread downstream to waterbodies in western Arnhem Land and KNP (Storrs & Julien 1996). In August 1990, the weed jumped catchments and appeared at Danbandji on Nourlangie Creek, a tributary of the South Alligator River. Again attempts at eradication failed and the infestation spread to surrounding areas and further downstream during the wet season of 1992/93. By the following wet season it was being flushed into the main South Alligator River (Storrs & Julien 1996).

2.4 Conceptual models

Conceptual models, based on known information on mimosa, para grass and salvinia and the potential ecological, cultural and socio-economic impacts are shown in figures 2, 3 and 4.

2.4.1 Mimosa

Pressure: *Mimosa pigra*

↓

Major exposure pathways: Water, wind, vehicles, boats, stock, wildlife, feral animals, deliberate movements of earth and propagules

↓

Favoured wetland habitats: Floodplains, freshwater ponds and swamps

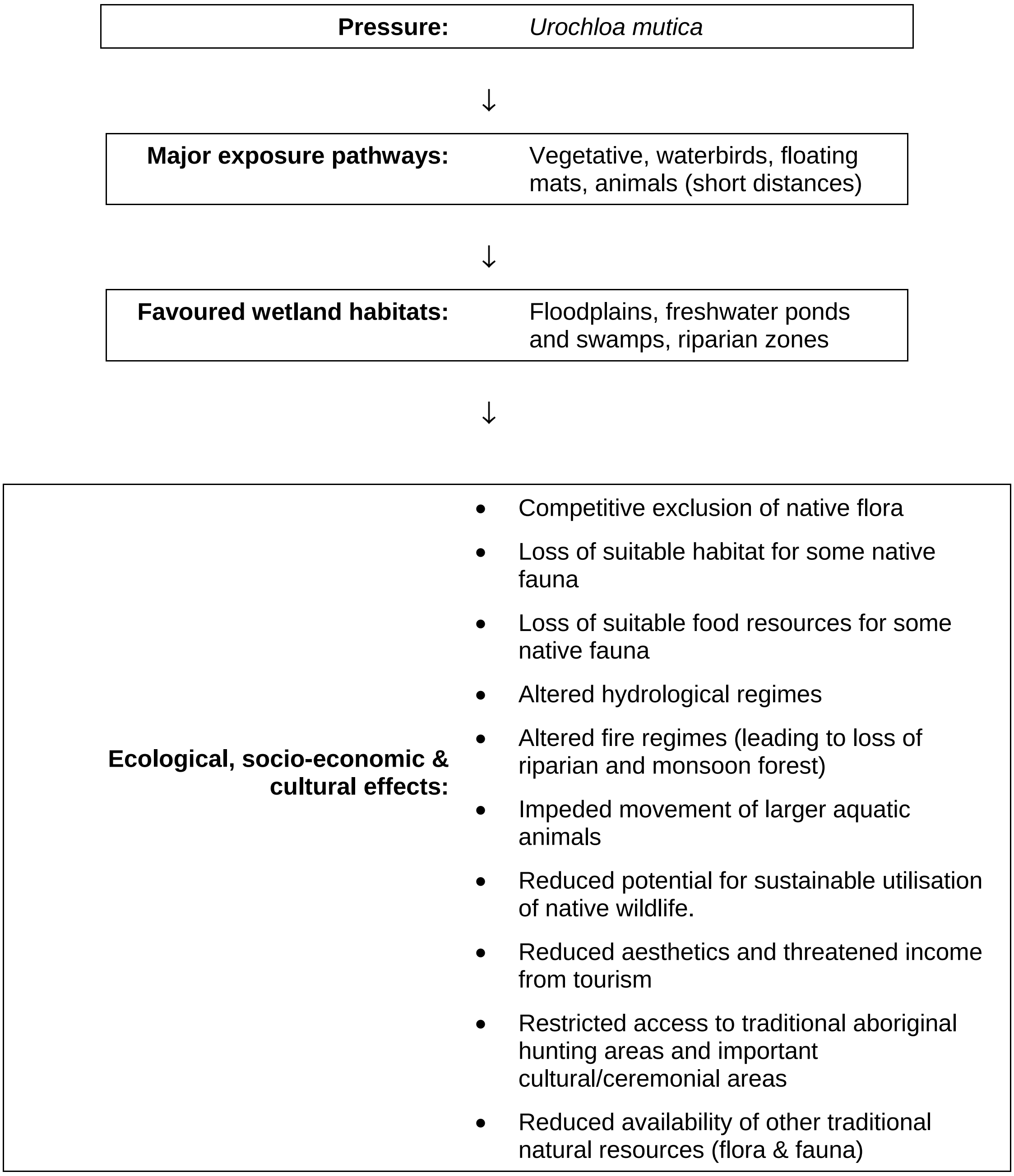
↓

Ecological, socio-economic & cultural effects:

- Competitive exclusion of native flora
- Loss of suitable habitat for some native fauna
- Creation of suitable habitat for some native fauna
- Loss of suitable food resources for native fauna
- Alteration of hydrological regimes
- Decreased capacity to manage vertebrate pests
- Competition with pasture grasses
- Reduced development, and increased production costs of pastoral and agricultural enterprises
- Reduced potential for sustainable utilisation of native wildlife.
- Diminished aesthetics and threatened income from tourism
- Reduced access to recreational fishing
- Restricted access to traditional Aboriginal hunting areas and important cultural/ceremonial areas
- Reduced availability of other traditional natural resources (flora & fauna)
- Diminished status as a nationally or internationally important wetland

Figure 2 Conceptual model of *Mimosa pigra*

2.4.2 Para grass



Pressure: *Urochloa mutica*

↓

Major exposure pathways: Vegetative, waterbirds, floating mats, animals (short distances)

↓

Favoured wetland habitats: Floodplains, freshwater ponds and swamps, riparian zones

↓

Ecological, socio-economic & cultural effects:

- Competitive exclusion of native flora
- Loss of suitable habitat for some native fauna
- Loss of suitable food resources for some native fauna
- Altered hydrological regimes
- Altered fire regimes (leading to loss of riparian and monsoon forest)
- Impeded movement of larger aquatic animals
- Reduced potential for sustainable utilisation of native wildlife.
- Reduced aesthetics and threatened income from tourism
- Restricted access to traditional aboriginal hunting areas and important cultural/ceremonial areas
- Reduced availability of other traditional natural resources (flora & fauna)

Figure 3 Conceptual model of *Urochloa mutica*

2.4.3 Salvinia

Pressure: *Salvinia molesta*

↓

Major exposure pathways: Water, floating mats, vehicles, boats, animals (short distances)

↓

Favoured wetland habitats: Floodplains, freshwater ponds and swamps

↓

Ecological, socio-economic & cultural effects:

- Competitive exclusion of native aquatic flora
- Loss of suitable habitat for some native fauna
- Loss of suitable food resources for some native fauna
- Altered hydrological regimes
- Altered nutrient status and water quality
- Restricted access to waterways (humans and wildlife)
- Reduced potential for sustainable utilisation of native wildlife.
- Reduced aesthetics and threatened income from tourism (particularly when areas are quarantined)
- Restricted access to traditional aboriginal hunting areas
- Reduced availability of other traditional natural resources (flora & fauna)

Figure 4 Conceptual model of *Salvinia molesta*

3 The potential effects

3.1 Effects on ecosystems

3.1.1 Mimosa

Mimosa is an enormous problem for wetland conservation. Largely intact natural landscapes can be completely altered, with floodplains and swamp forests being covered by dense monospecific stands of mimosa, which have little understorey except for mimosa seedlings and suckers (Braithwaite et al 1989). The severity of the impact of mimosa results from the following: (1) the high dominance by the invading species; (2) the gross change in vegetation structure; and (3) the conversion of a wide range of structural types of vegetation to a homogeneous tall shrubland (Braithwaite et al 1989).

Due to mimosa's ability to sprout adventitious roots, it may even modify waterbodies. In the dry season, seedlings establish along the receding waterlines (Lonsdale and Abrecht 1989), and with the inundation of the next wet season, the natural water flows are reduced by the adventitious roots, resulting in increased sediment deposition. Thus, the thickets are able to continue their advance until only a tiny remnant of open water remains in the deepest parts.

3.1.2 Para grass

Like mimosa, para grass also dominates other vegetation and forms a monospecific habitat. As the density of para grass is usually greater than that of native vegetation, hydrological regimes may be altered resulting in reduced flows and greater deposition of sediment. This in turn could also lead to shoaling or elimination of waterbodies.

3.1.3 Salvinia

Salvinia's main ecosystem impacts occur as a result of the dense mats that often completely blanket waterbodies and fringing vegetation. These mats are sometimes invaded by other plant species that colonise and stabilise the mats to form extensive floating communities (Finlayson et al 1988). Light penetration into the water column can be almost non-existent under a dense mat leading to reduced algae and macrophyte growth. Water quality under the mats is affected, having lower oxygen levels, higher carbon dioxide and hydrogen sulphide concentrations and being more acidic and warmer than nearby open water (Mitchell 1978). The nutrient status of the water is altered as salvinia stores large amounts of nitrogen and phosphorus. The maximum rate of nitrogen uptake has been calculated at about 6000 kg per hectare per year (Room 1986).

3.2 Effects on native flora

3.2.1 Mimosa

Unless cited otherwise, the following information on effects of mimosa to native flora and fauna is summarised from Braithwaite et al (1989), who investigated two study areas. One on the Adelaide River (6 sites) where the infestation was about five years old and relatively stable, and one on the Finniss River (8 sites) where the infestation was about three years old and still in the process of invasion.

Once established, mimosa is able to out-compete native herbaceous layer vegetation for light moisture and nutrients, although the relative importance of these three factors has not been determined. A comparison of incident light measurements beneath the mimosa canopy found that the sedge-land sites, which carry no trees, received 100% of the sunlight in the absence of mimosa, but only 62% (Finniss River sites) and 81% (Adelaide River sites) when it was present.

The *Melaleuca* dominated swamp forests fringing the floodplains have a rather open canopy and mimosa has also penetrated this habitat, preventing seedlings of the native forest trees from establishing. Incident light measurements show that although 75% passed through the native tree canopy, only 26% reached the ground flora with the additional presence of a mimosa canopy. Due to the demonstrated exclusion of native tree seedlings, it is proposed that the mature native tree canopy would eventually die out, and these swamp forests, like the sedgelands would become mimosa-dominated shrubland.

The results indicate that the effect on the light regime at ground level, regardless of competition for moisture and nutrients, may be sufficient to account for the observed reduction in the number of tree seedlings, biomass and species diversity of the herbaceous layer. The light measurements were taken during the dry season when the weed has a relatively sparse canopy. The impacts could possibly be exacerbated in the wet season, when the denser canopy of a lush mimosa thicket may prevent around 90% of the incident light from reaching the ground.

Cook (1992) compared vegetation sites with and without mimosa for three different communities on the Oenpelli floodplain. Where mimosa was present, the projected cover of native herbaceous species was less than one third and the species diversity was less than one half of those where mimosa was absent for floodplain margin and back-swamp communities. In the open floodplain communities, the cover of understorey species was similar and species richness was only slightly less where mimosa shrubs were present.

Studies and observations show that herbaceous vegetation does recolonise following the removal of mimosa (Cook 1992, Searle & Fell 2000, DPIF 1997). At the site of the Oenpelli infestation, two years after the removal of mimosa, the diversity of herbaceous species had returned to levels similar to those found in the absence of mimosa. However, the actual cover of these species did not respond as rapidly as the diversity, and remained well below that found in areas yet to be invaded by mimosa (Cook 1992). Field observation has shown significant differences in the recolonisation rate of native vegetation between wetland areas. Recolonisation of native vegetation was very rapid within the Finniss River floodplain, occurring within two years. In contrast, recolonisation of native floodplain vegetation in some areas of the Daly River has been very poor and remained limited for four years following mimosa control. The degree of recolonisation of herbaceous species is dependent upon a variety of factors including, but not limited to, the amount of native seed importation, native seed soil stores, rainfall and inundation events, the effects of onsite ecological disturbance such as fire or vegetation removal from animal grazing or trampling, and the accuracy of herbicide application rates in relation to different soil types and hydrology. Over-application can cause soil scalds, where very little vegetation is able to establish (Cook 1996). An improved understanding of the recolonisation process following mimosa control is critical to achieving sustainable and long-term management of mimosa by limiting future mimosa seed germination and limiting seedling growth and development.

3.2.2 Para grass

Two studies investigated the effects of para grass on native flora on the Magela floodplain (Knerr 1998, Douglas et al 2001). Knerr (1998) examined four Magela floodplain grassland communities (*Urochloa mutica, Hymenachne acutigluma, Oryza meridionalis* and *Pseudoraphis spinescens*) for species richness and diversity. Combined data for wet and dry seasons showed that *Urochloa* grassland had the lowest species richness of the four communities. However, this was not significantly different ($p>0.05$) to the species richness of the *Pseudoraphis* grassland. The *Urochloa* grassland had the lowest species diversity (combined seasons) and this was significantly different ($p<0.05$) to the species diversity of the other communities. Species richness did not change in the *Urochloa* grassland between seasons, in contrast to all other communities which showed significant changes. Although species diversity increased in the *Urochloa* grassland in the wet season, this was still much lower than the diversity found in other communities.

An increase in area of the *Urochloa* community coupled with a corresponding decrease in area of the *Oryza* community from 1991 to 1996 suggests strongly that para grass has invaded the latter community. *Hymenachne* grassland appeared not to have been invaded during this time, although it is thought that this may be occurring but over a longer time scale. The *Oryza* community is perhaps more susceptible to invasion as it is the only annual of the four grass species and, therefore, relies entirely on the seed bank to persist over the dry season (Knerr 1998). It is during this dormant stage that perennials such as para grass can establish in *Oryza* habitats.

Douglas et al (2001) also examined the effects of para grass on Magela floodplain grassland communities (*Urochloa mutica*, *Hymenachne acutigluma, Oryza meridionalis* and a herbicide treated *U. mutica* grassland). Plant species richness and cover were measured for both wet and dry seasons, plant biomass and surface area were measured during the wet season, and rates of breakdown and dry season fuel loads were also investigated. In the dry season, para grass was the only vegetation type that occurred in monospecific stands. The structurally similar *Hymenachne* community contained five plant species, while the rice and sprayed patches each had a total of 11 species. Species richness increased during the wet season and a pattern similar to that of the dry season was evident, where total richness was highest in the rice and sprayed habitats and lowest in the para grass habitat. The lower species richness was not as marked at the quadrat scale, however total richness (summed across all quadrats) was much lower in para grass, indicating a major reduction in plant species richness at larger spatial scales.

Para grass had a higher biomass than all of the other communities for both the wet and dry seasons. Its invasion of other grass habitats leads to greater fuel loads, which would most likely increase the intensity and extent of floodplain fires, particularly in the late dry season. On the Magela floodplain there is already possible evidence of fire impacts with dead patches of badly fire scarred *Melaleuca* trees with a para grass understorey (M Douglas pers obs). Parks Australia North staff have also reported that hot para grass fires have been responsible for damage to some monsoon vine forest patches (P Barrow pers comm.).

3.2.3 Salvinia

The ecosystem impacts of altered light regimes, water quality and nutrient levels as outlined in section 3.1.3 will ultimately affect the growth, condition and species composition of the aquatic flora. Quantitative information on these effects is currently unavailable.

3.3 Effects on native fauna

3.3.1 Mimosa

Effects on native fauna result from the dramatic floristic and hydrological changes brought about by mimosa invasion. The structural change from sedgeland to tall shrubland has a more severe effect on the fauna than the clearing of native forest to make way for introduced pine plantations (Friend 1980). Braithwaite et al (1989) identified a number of species that were affected both adversely and favourably by mimosa invasion at their Adelaide River and Finniss River study sites. Using these data they were also able to hypothesise on the general effects of mimosa on patterns of animal abundance and diversity in these and other areas.

As other environmental factors in addition to mimosa appeared to act on the abundance of some species, multiple regression analysis was used to test 128 combinations of habitat variables to examine their influence.

3.3.1.1 Birds

The sedgeland-dwelling *Cisticola* spp. and the wet forest species *Conopophila albogularis* (rufous-banded honeyeater) and *Geopelia striata* (peaceful dove) exhibited lower abundance on sites invaded by mimosa, whilst the willie wagtail (*Rhipidura leucophrys*) appeared to increase in abundance. The abundance of terrestrial birds was positively related to both foliage height diversity and herbaceous biomass, and to woody species diversity. The species richness of terrestrial birds was positively correlated to both woody and herbaceous species diversities. This phenomenon was also observed on the Oenpelli floodplain where the disappearance of many species of ground-feeding birds and the appearance of birds with other forage zones, was clearly due to the conversion of the native-grass- and sedge-land to a mimosa shrubland. The increase in diversity of terrestrial species was probably associated with the increased cover and nesting sites provided by the mimosa. The favourable effects of increased cover for terrestrial species were clearly demonstrated by the disappearance of the raptors *Tyto alba, Aquila audax,* and *Elanus notatus* from areas with dense mimosa, despite the presence of an abundance of their prey species (Cook 1992).

Waterbird abundance was negatively correlated to river system and to woody species diversity. Waterbird richness related to foliage height diversity and positively to herbaceous diversity and root height. Treeless, species-rich, deep-water sedgeland is the prime habitat for waterbird populations, which rely on it for breeding and feeding. Further loss of this habitat through mimosa invasion would see an increasing negative impact on waterbird populations.

Some species such as the magpie goose (*Anseranas semipalmata*) and the brolga (*Grus rubicundus*) have either disappeared or are now much less common in other parts of Australia, increasingly using the wetlands of Northern Australia as a refuge (Frith & Davies 1961, Blakers et al 1984). Indeed 60–70% of the total population of magpie geese in northern Australia seek refuge in two or three areas in Kakadu National Park (KNP) towards the late dry season (Bayliss & Yeomans 1990). The floodplains of the Adelaide and Mary Rivers encompass the most important nesting habitat in the NT for magpie geese, accounting for 32–52% of the total number of nests between 1984 and 1986 (Bayliss & Yeomans 1990).

The main rookery sites for species such as the sacred ibis (*Threskiornis aethiopica*), royal spoonbill (*Platalea regia*) and little pied cormorant (*Phalacrocorax melanoleucos*), and the main roosting and nesting sites of most of the raptors are found in the wet forests (paperbark, riparian and monsoon). As for the sedgelands, destruction of these habitats would impact greatly on these and other similar bird species.

3.3.1.2 Mammals

Small mammals seemed to favour the dense mimosa canopy. The Dusky Plains rat (*Rattus colletti*) greatly favoured the Adelaide River mimosa sites, whilst the small insectivorous dasyurid (*Sminthopsis virginiae*) was particularly abundant in the Finniss River mimosa sites, with all but one of the 28 captures being in the two mimosa – no paperbark sites. Analyses showed that mammal abundance was positively correlated to mimosa cover/abundance and negatively correlated to woody species diversity. Mammal species richness was positively correlated to mimosa cover/abundance and negatively correlated to river system and herbaceous species diversity.

The prime attraction of mimosa for small mammals is likely to be a favourable microclimate or protection from predation. The prickly canopy and dense skirts of adventitious roots at the base of mimosa stems provide ideal shelter and protection from avian predation (Braithwaite & Lonsdale 1987). It is thought that these small mammals will probably survive only as long as the mimosa occurs in patches from which they can make forays into the surrounding sedgelands for food. If mimosa takes over all the sedgelands, the area will probably be unable to support the increased population (Lonsdale & Braithwaite 1988).

Large groups of flying-foxes (*Pteropus alecto* and *P. scapulatus*) roost and feed in the wet forests for much of the year (Friend & Braithwaite 1985). Severe destruction of this habitat as a result of mimosa invasion could potentially cause a decline in flying-fox populations. Flying -foxes are important as major pollinators and seed dispersers for trees throughout northern Australia, and a source of food for Indigenous people.

3.3.1.3 Reptiles and Amphibians

The majority of the reptiles captured during the study were skinkid lizards. The skinks *Cryptoblepharus plagiocephalus* and *Carlia gracilis* decreased in abundance at mimosa infested sites. The skinkid lizards are rarely found on the floodplains, preferring the forests and woodlands. Mimosa, however, appeared to provide an unsatisfactory microhabitat and few lizards were found in the mimosa-dominated areas.

Amphibians showed no distinct pattern with respect to mimosa. Abundance was positively correlated to mimosa cover/abundance, foliage height diversity and herbaceous biomass. Species richness showed some negative relationships with mimosa cover/abundance and root height, and positive relationships with foliage height diversity and buffalo usage. Species richness is not unexpectedly correlated to the wetness of the site. The results were probably also affected by the difference in time of year of the sampling of the two river systems and detectability in ground vegetation of different density.

3.3.2 Para grass

Douglas et al (2001) also examined the effects of para grass on aquatic macroinvertebrates, terrestrial invertebrates and fish communities on the Magela floodplain. For the aquatic macroinvertebrates they concluded that para grass invasion had little or no impact. Richness was generally lower in the *Hymenachne* sites, probably due to the anoxic nature of the benthic zone in this habitat. There was evidence of adverse effects of para grass on terrestrial invertebrates in the dry season, but only when para grass displaced *Hymenachne*. Total richness and abundance of terrestrial invertebrates were lower in para grass than in *Hymenachne*, thus widespread displacement of the native floodplain vegetation could reduce the biodiversity and abundance of this fauna. Results for the wet season showed the potential (if para grass replaces *Hymenachne*) for a reduction in terrestrial invertebrate biodiversity. n contrast to the dry season results, however, invertebrate abundance did not appear to decline.

There were no significant differences in the number of fish species or total fish abundance among the four vegetation types. Nevertheless, Douglas et al (2001) did suggest, however, that the absence of adverse effects may have been because the infestation conditions at the time of sampling still represented a mosaic of vegetation communities. If para grass infestation was very widespread and forming dense monocultures, resulting in a much lower structural diversity, then some reduction in diversity and/or abundance of fishes and fish communities could be expected.

The greater wet season biomass of para grass corresponds to a greater density of vegetation in the water column and this may impede the movement of larger aquatic animals such as turtles and water monitors (Douglas et al 2001). The greater dry season biomass and subsequent hotter fires could also threaten species that aestivate (Douglas et al 2001) or even take refuge in the cracking clays during the heat of the day. Examples of such species could include turtles (*Chelodina rugosa*), small mammals (eg *Rattus colletti*), amphibians (eg *Litoria dahlii*), invertebrates such as Prosobranch snails (eg *Bithyniidae* spp., *Viviparidae* spp.), crabs (*Holthuisana* spp.) and microcrustacea (eg cladocerans, copepods & concostrachans) (C Humphrey pers comm, Julli 1986).

The greatest impacts on native fauna will most likely result from a reduction in the diversity, distribution and abundance of plant species important for food resources. In particular, wild rice is perhaps the most important food resource for many wildlife species and happens to be the species most frequently replaced by para grass on the Magela floodplain (Knerr 1998, Cowie & Werner 1993). It is a prolific producer of seeds high in energy and protein, essential for pre-fledging magpie geese which need to grow quickly in time to flee breeding swamps before they dry (Frith & Davies 1961, Whitehead et al 1990, Whitehead and Dawson 2000). The bulbs of the sedge *Eleocharis* spp, which are more common in wild rice communities than in para grass monocultures (Douglas et al 2001), are another important food source for magpie geese (Frith & Davies 1961, Corbett et al 1996). Para grass cannot support the growth rates obtained from a diet of these native species (Whitehead and Dawson 2000). Magpie geese may also be detrimentally affected by para grass invasion as they preferentially nest in *Eleocharis*/*Oryza* (Corbett et al 1996) communities. Some other waterbird species that visit the floodplains to feed on wild rice seeds include the Grass Whistling duck (*Dendrocygna eytoni*), *Brolga* (*Grus rubicundus*), Little Whimbrel (*Numenius minutes*) and Little Corella (*Cacatua sanguinea*) (Whitehead 1992).

Native grasses have also been shown to be an important source of seeds for granivorous rodents such as the Dusky Plains rat (*Rattus colletti*). Wild rice and *Eleocharis dulcis* both form an important component of the diet of this species (Wurm 1998).

3.3.3 Salvinia

There is no published information that quantifies the effects of salvinia on fauna. Anecdotal evidence from Kakadu National Park suggests that the numbers and species of waterbirds were fewer on billabongs with salvinia infestations (M Storrs & M Julien pers obs. 1991–1994, B Hall pers comm. 1994). There is evidence of declines in numbers of small fish, such as catfish (*Neosilurus* spp.) in salvinia-infested billabongs and a decline in the population of file snakes (*Acrochordus arafurae*) in a severely infested billabong has been reported (T Madsen, pers comm. 1994) (Storrs & Julien 1996).

4 The potential extent

4.1 Current distribution

Information on the distribution of the three weed species has been obtained for the Magela floodplain (see section 1.5.2.2). Not all data sources are represented on the maps because in most cases there was almost complete overlap with the data sources that *are* represented. Additionally, there are number of weed locations that are outside of the coverage of the current IKONOS® swath used in the maps. All known weed locations for the catchment will be presented in the final report (see section 1.5.2.3). The 1987 data was collected without the aid of a GPS, so the exact locations on the maps cannot be guaranteed. It is quite likely that the overlapping sites in figure 6 are actually the same site.

4.1.1 Mimosa

Although figures 4 and 5 show a number of mimosa plots, there are no current mimosa infestations on the Magela floodplain, as all infestations were located and treated before they became a serious problem. The maps represent a history of mimosa invasion on the floodplain, the site discovery dates spanning some 15 years. This emphasises the need for constant surveillance as seeds are imported from outside areas or have been distributed from prior existing plots on the floodplain. The longevity of the seeds in the soil (see sections 2.1.3 & 2.1.4) also justifies long-term surveillance.

For the purposes of this report, the mimosa map (figures 4 and 5) has been divided into the upper and lower sections of the floodplain. The IKONOS® swath is very long and quite narrow, and when the map was projected on a single page, the details of the floodplain features adjacent to the plots were difficult to distinguish.

4.1.2 Para grass

Figures 6 and 7 show the distribution of para grass on the Magela floodplain using three different data sources. This map has also been divided into to two parts to enhance the detail. A more detailed survey needs to be conducted to accurately map the true extent of Para grass on the floodplain. The new locations recorded in March 2003 were obtained during a field trip that was also concerned with collecting data for detailed vegetation mapping, thus the time allocated exclusively for weed survey was limited. The points represent GPS spot locations recorded whilst travelling in an airboat.

Where there are a series of points closely gathered along a path, it is usually the case that para grass is continuous along the path. The path to the east of the existing (Knerr 1998) infestation is along the current airboat track, so it is possible that these boats may be assisting the spread of this weed. Where the new spot locations are close to the existing infestation, it is possible that, in some cases, these points now represent the outer boundary of the coverage (ie the existing infestation has spread out to this point and occupies all the area between the previous infestation and the new location). Again, this phenomenon could be determined with a more detailed survey, perhaps in the early dry season when the weed is growing vigorously, is fully exposed and has not been affected by fire. This would also be the optimum time to determine the distribution of para grass using high resolution yet cost-effective remote sensing captures (eg QuickBird™) given that there is less cloud cover than in the wet season and less atmospheric noise (eg smoke) than in the late dry season.

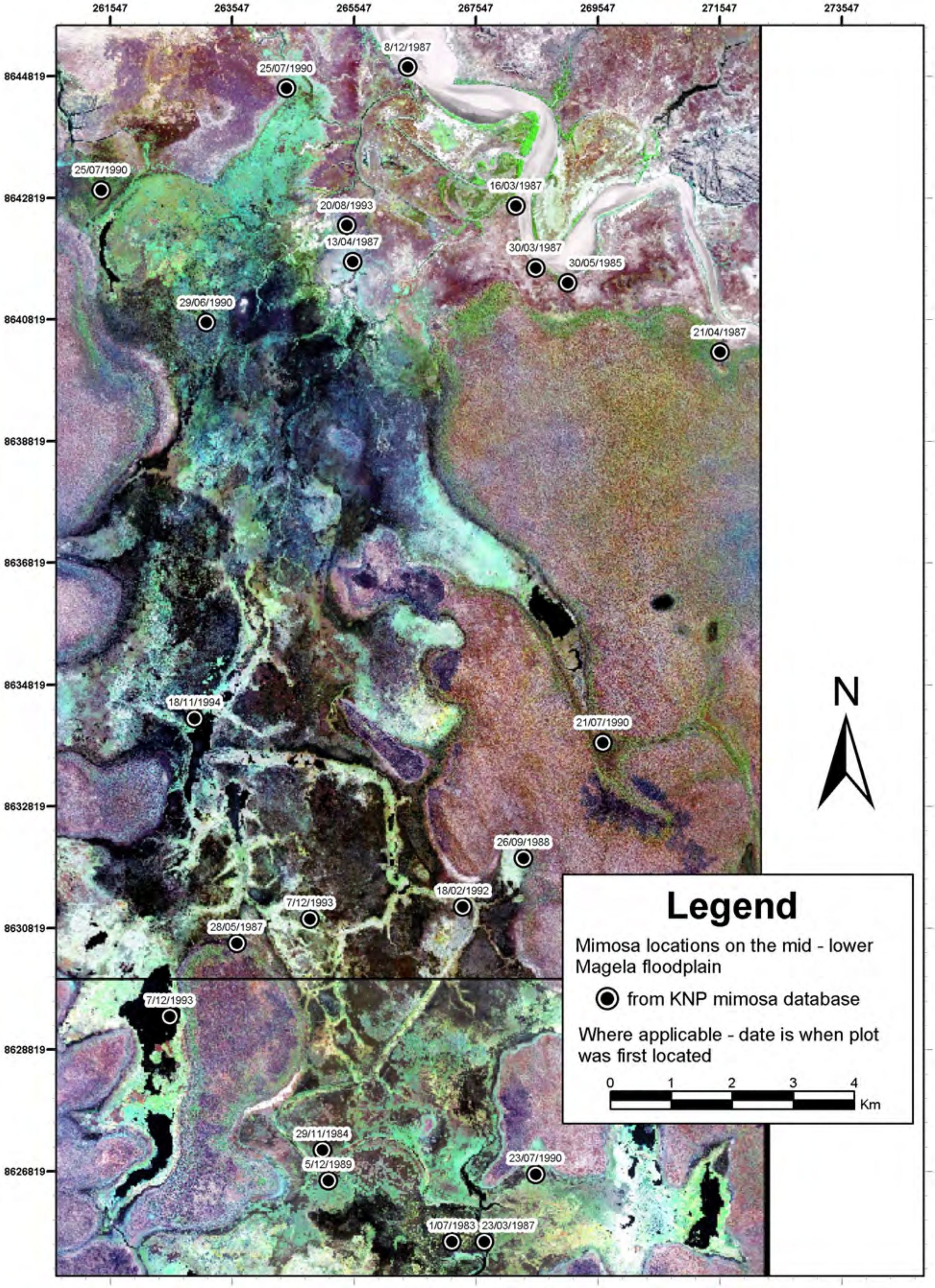


Figure 5 Mimosa locations on the lower section of the Magela floodplain

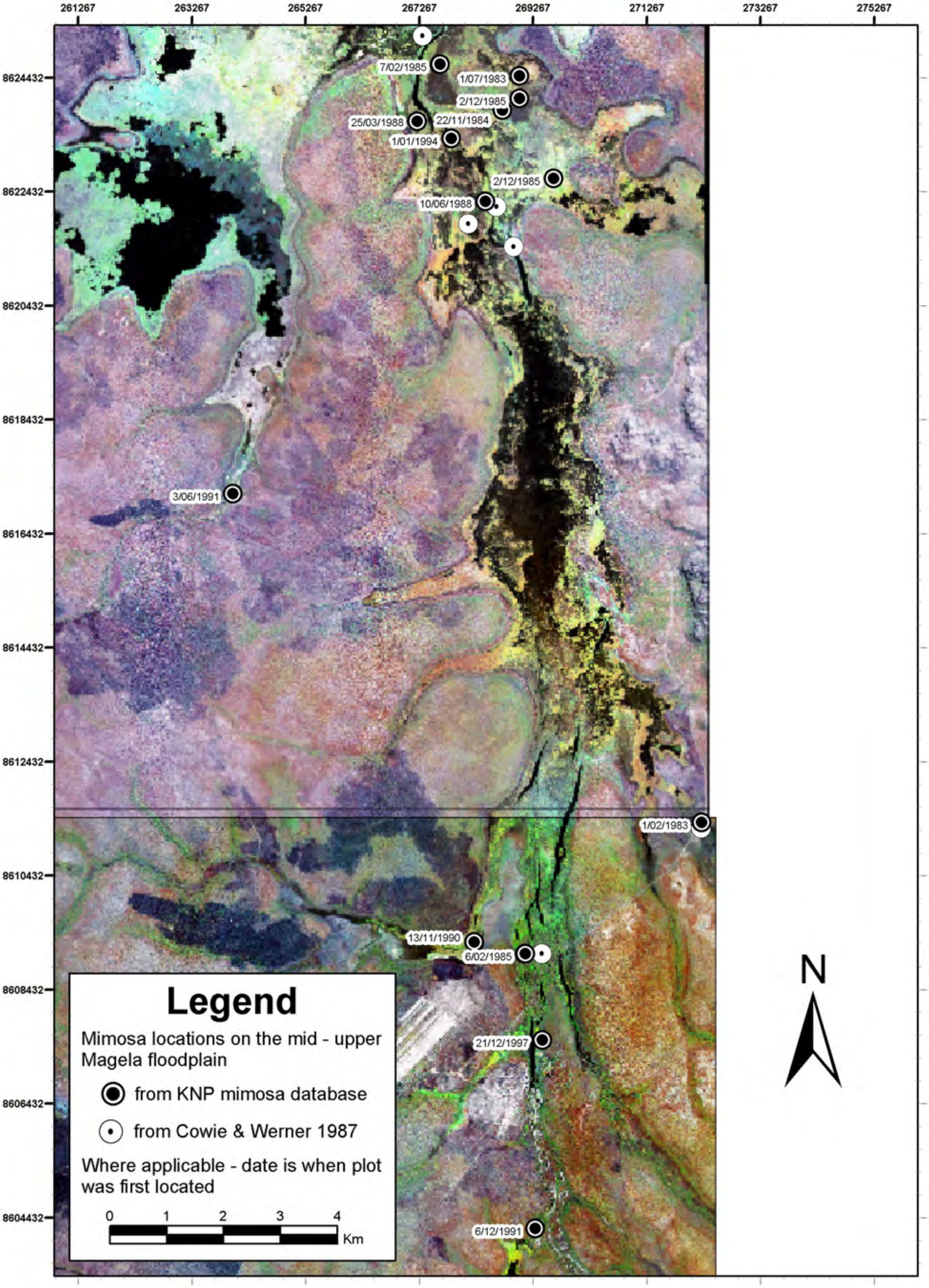


Figure 6 Mimosa locations on the upper section of the Magela floodplain

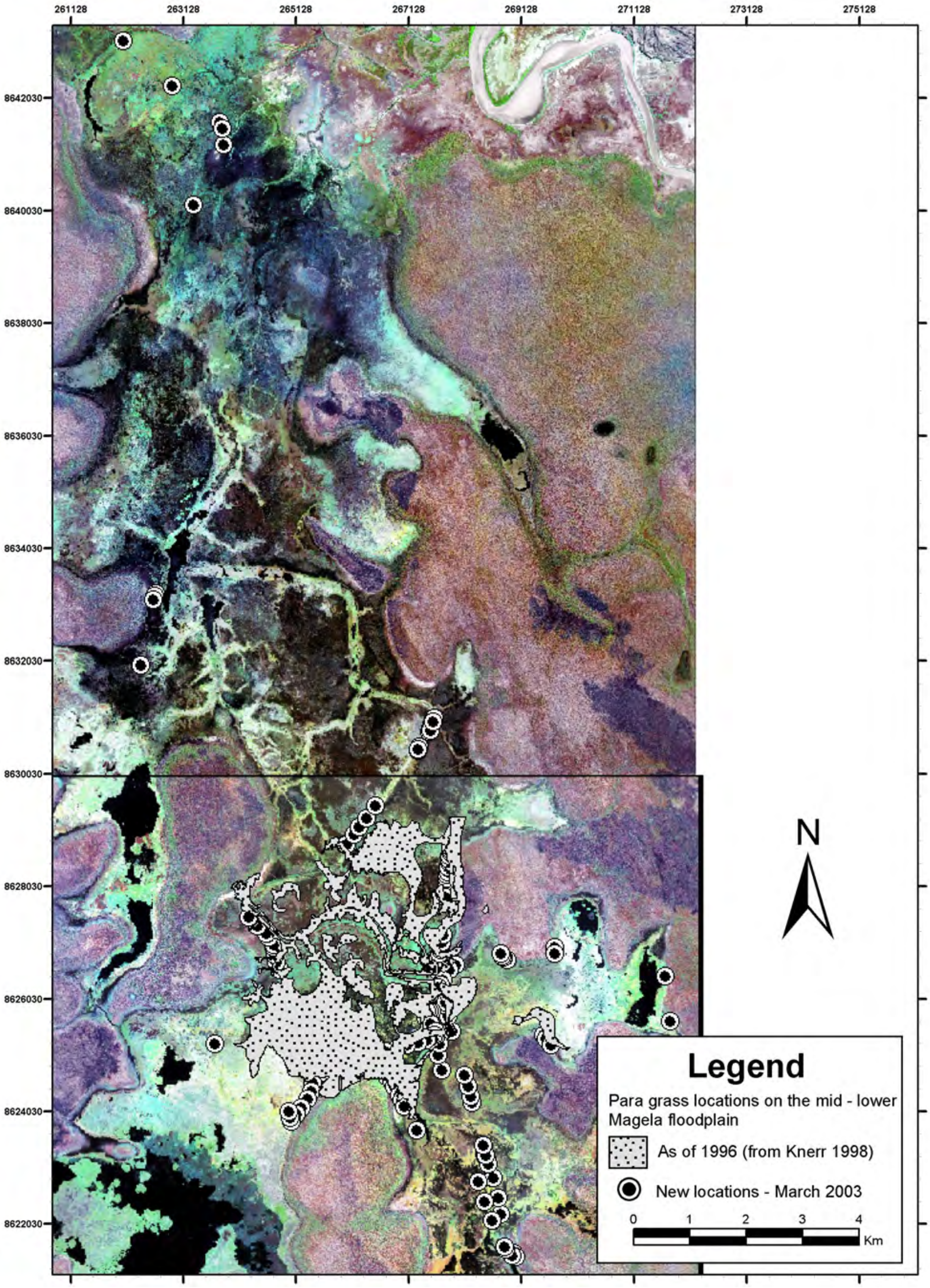


Figure 7 Para grass locations on the lower section of the Magela floodplain

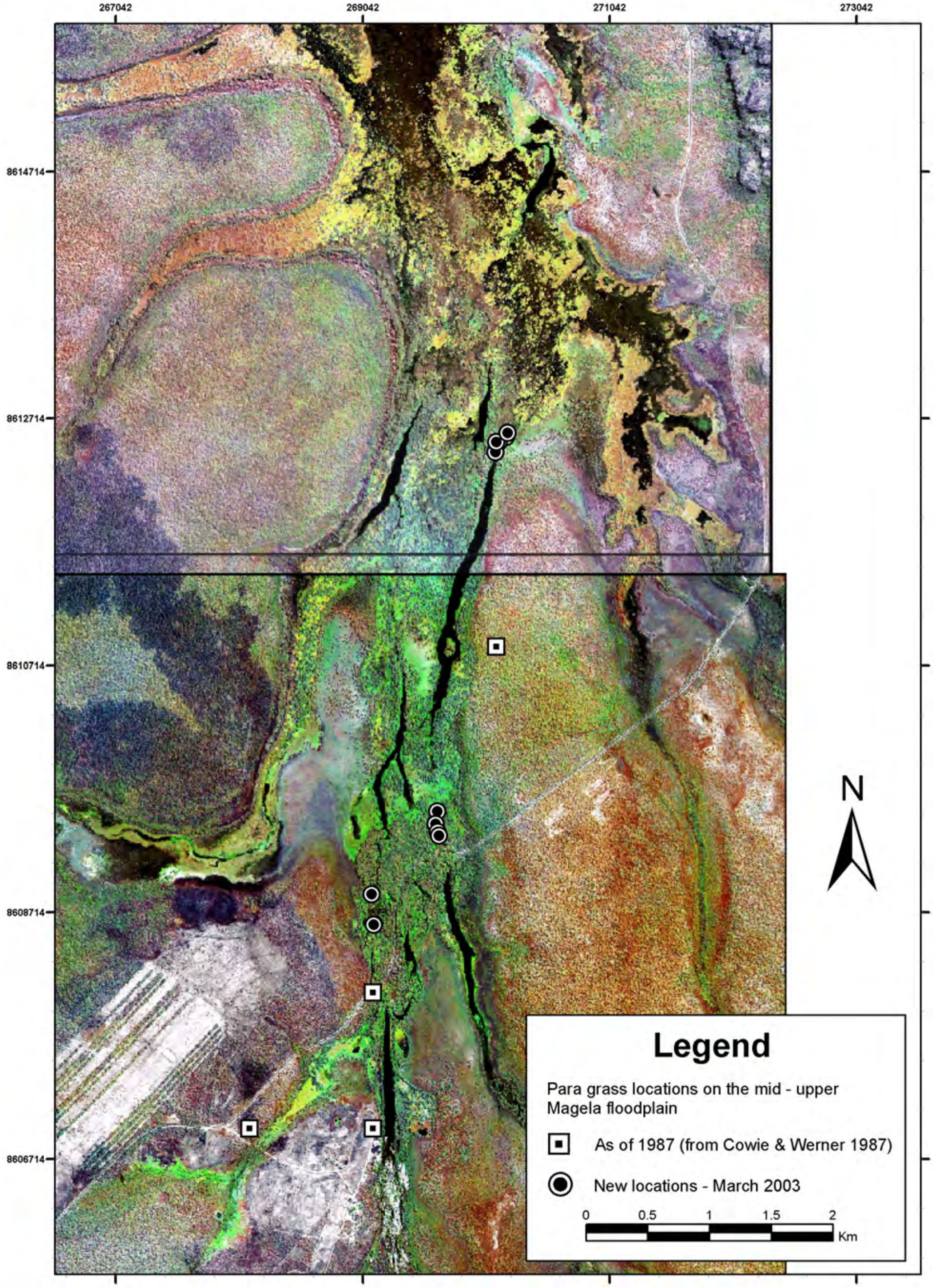


Figure 8 Para grass locations on the upper section of the Magela floodplain

4.1.3 Salvinia

(adapted from Storrs & Julien (1996))

The current distribution of salvinia on the Magela encompasses the entire Magela floodplain. Such was the status of the weed only two or three years after it was first discovered in 1983 (Finlayson et al 1994). A few years after this it spread upstream of its original source location and invaded Mudginberri Billabong and the surrounding swamplands (Storrs & Julien 1996).

The dynamic nature of salvinia means that the actual coverage within the distribution varies considerably. The mats on the billabongs rarely persist for a complete annual cycle due to damage caused by the introduced weevil and wet season flushing (Finlayson et al 1994). In the late wet season, the plant is at its most vigorous and can completely cover a billabong. Toward the late dry season, the salvinia mats generally show dramatic symptoms of weevil damage and sink prior to the wet season floods (Storrs & Julien 1996, Julien & Storrs 1994), usually in December or January. The most significant damage is caused by the larvae as they burrow through the stem, destroying the root–shoot link (Sands et al 1983, Julien et al 1987) causing the plant to become friable and waterlogged, and to eventually sink (Forno et al 1983). The strength of the wind and the subsequent degree of compaction of the mats can also greatly influence the area of salvinia cover on a waterbody (Storrs & Julien 1996).

In some years little salvinia growth occurs and, in contrast, in other years the biomass can increase rapidly often resulting in 100% coverage of the billabongs. The rate of salvinia biomass accumulation, and the actual amount of growth, is influenced by factors that are linked to the population dynamics of the weevil. The timing of the onset of the wet season rains and the timing and size of the main flood events appear to be the key variables. The initial flood of the wet season provides an influx of nutrients generating a high potential for growth of salvinia. A 'good' wet season has follow-up floods that dilute the wetland system, markedly reducing nutrient levels, thus reducing salvinia's growth potential. In a 'poor' wet season major floods are slow to arrive, allowing a 'soup' of nutrient rich waters to enter the waterbodies thus leading to a high growth potential. If weevil numbers are low, this high growth potential is met. Naturally occurring fish kills (Bishop 1980, Bishop et al 1982), often involving many thousands of fish, also inject extra nutrients into a waterbody which can increase the potential growth rate and biomass of salvinia. Storrs & Julien (1996) found that with the early arrival of major floods and associated flushing effect, combined with an absence of a decline in weevil numbers, generally meant that weevils were able to restrict the lower growth potential of salvinia. Both the weed and weevil populations increased but salvinia growth rates were suppressed and biomass and cover restricted, as a result of the relative abundance of the weevil. They also found that in the 'poor' wet season scenario, the salvinia grew rapidly for some time before any flushing occurred. At this time weevil numbers were very low and could not keep pace with the salvinia growth despite an increase in their numbers. Weevil numbers were found to be low because of population crashes that occurred late in the wet season. At this time, salvinia was severely damaged by weevil populations that had increased so rapidly that they had consumed virtually all of their food source.

Following the monitoring recommendations of Storrs and Julien (1996 p52), Ross Salau of Parks Australia North has a photographic record of selected billabongs on the Magela and Nourlangie systems. Photos were taken approximately every 6–8 weeks from the ground and from the air. The length of records varies but encompassed most of the 1990's decade and up to about 2001 depending on the site. Figure 8 shows salvinia on Jabiluka billabong (Magela) at various times of the year in 1992 and 1993.

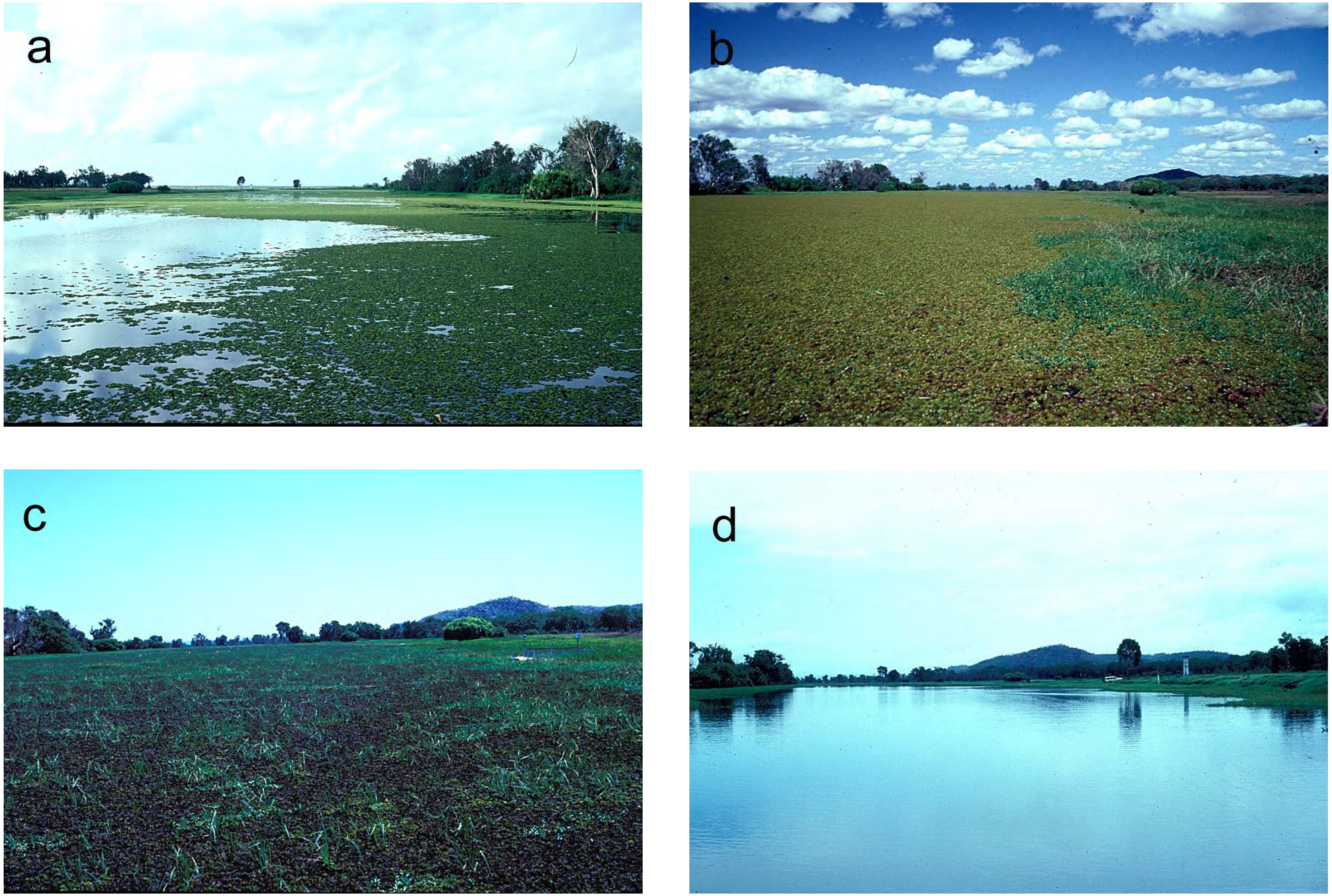


Figure 9 Salvinia on Jabiluka billabong in 1992/93 – 9a. salvinia is growing rapidly into the secondary form and is starting to form vast mats; 9b. July, thick mats have covered the entire billabong for some time, the yellowing is a result of nutrient deficiency and weevil damage. 9c. October, the brown colour is salvinia killed by severe weevil damage, just prior to it sinking. 9d. January 1993 a 'clean' billabong again after the mats have sunk and the wet season floods commenced (Photographs courtesy of Ross Salau).

5 Bioeconomic modelling

The three basic aims of population management are control, sustainable harvesting and conservation. Regardless of the aim, however, there must be a good understanding (or model) of the dynamics of the population in order to manage them. Additionally, management decisions must be linked also to socio-economic considerations or, put simply, the associated benefits and costs of any action or activity. Bioeconomic models help population managers bridge the gap between knowledge of the biology of the species being managed and knowledge of socio-economic consequences. Such models provide a powerful strategic framework, or Decision Support Tool (DST), for effective management of any natural resource. With respect to the management of invasive species, the most useful bioeconomic model will encompass the following three key sub-models (or functions) which are explicitly linked: (1) a damage-abundance function (or suite of functions if multiple damage exists); (2) a population dynamics model; and (3) a cost-of-control function.

5.1 Mimosa

5.1.1 Kakadu National Park

Information on the amount of effort expended to control patches of varying densities of mimosa across the park was extracted from the KNP mimosa database (table 1) which has records from 1981 to present. Figure 10 represents a ‘first pass’ effort-abundance curve over the 22 years of mimosa survey and control. Further information on labour and operational costs will be obtained to convert this function to a practical ‘cost-of-control curve’ so that managers can simulate and compare various control options. The curve follows the classic pattern where effort (days per plant) is high to treat a lower plant to plot ratio and decreases rapidly for higher plants/plot ratios. That is, it costs more resources (or takes more effort) to treat a smaller number of plants over a larger area than it does to treat a larger number of plants over a smaller area. If Kakadu were infested with large areas of mimosa, the model parameters could be set to determine the initial reduction and the subsequent maintenance control effort relative to a target density required each year *ad infinitum*. With the current mimosa situation in KNP, the control cost model can help managers minimise survey costs of existing and new sites based on the history of available data.

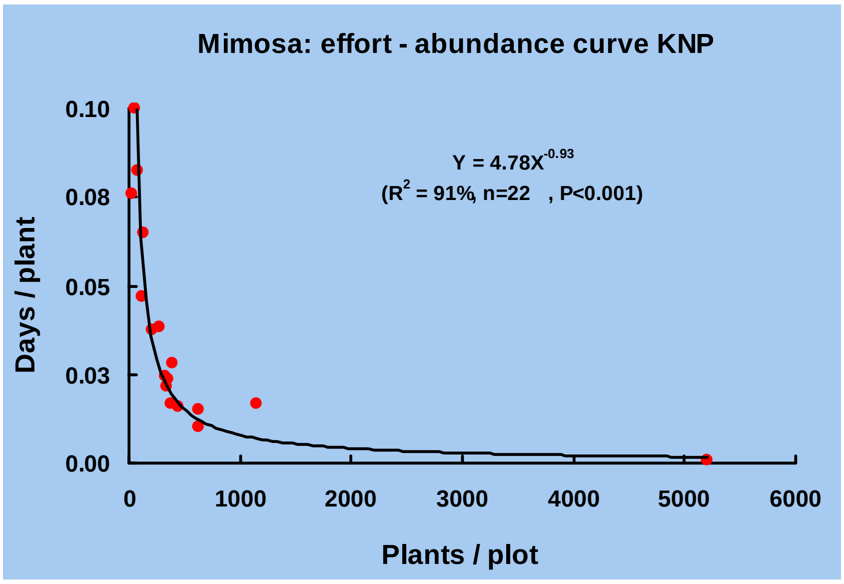


Figure 10 Control effort - abundance curve for mimosa survey and control in KNP

Table 1 Summarised mimosa control information for KNP

Year	No. of plots	Visits	Person (days)	Total plants	Days/plant	Plants/plot
1981	2	4	10.28	103	0.10	51.50
1982	1	5	10.7	15	0.71	15.00
1983	6	10	20.98	31197	0.00	5199.50
1984	17	152	326.34	19473	0.02	1145.47
1985	36	187	402.84	14275	0.03	396.53
1986	45	216	464.32	12163	0.04	270.29
1987	65	280	605.34	40363	0.01	620.97
1988	51	174	374.4	17250	0.02	338.24
1989	45	169	362.4	14859	0.02	330.20
1990	71	258	555.78	14789	0.04	208.30
1991	50	165	358.32	22440	0.02	448.80
1992	104	309	662.52	65316	0.01	628.04
1993	123	363	785.1	47206	0.02	383.79
1994	102	249	532.46	11329	0.05	111.07
1995	93	192	416.46	4048	0.10	43.53
1996	152	584	1277.08	54250	0.02	356.91
1997	162	663	1438.7	22138	0.06	136.65
1998	162	540	1135.28	9282	0.12	57.30
1999	167	473	1006.7	12228	0.08	73.22
2000	162	590	1252.06	10562	0.12	65.20
2001	167	446	930.7	3178	0.29	19.03
2002	38	41	85.4	1123	0.08	29.54

blank person entries in the original database are calculated as mean of total entries - across all plots for all years. 2002 is an incomplete year

5.1.2 Gunbalanya – demonstration of a Decision Support Tool for mimosa control

Cost-of-control information was extracted from the major mimosa control program (1992-1996) on the Gunbalanya floodplain to the east of KNP in order to complement the ecological risk assessment process for mimosa on the Magela floodplain. Mimosa on Gunbalanya floodplain represents a major source of recolonisation and, hence, risk to the Magela floodplain. The extent of mimosa on Gunbalanya in 1992, adjacent to World Heritage Kakadu National Park, justified the large investment of approximately $8 million for control. Historical data on control costs and spread rates of mimosa at Gunbalanya are sufficiently complete to construct a first cut 'proof-of-concept' bioeconomics model which weed managers can use as a Decision Support Tool (DST).

The scenario simulation model, outlined below, only combines the last two of the three key sub-models described above (population dynamics & cost-of-control sub-models). Both of these sub-models were derived from empirical field observations (see below). Although data exists to derive damage-abundance functions (Cook 1992 for the loss of floodplain plant biodiversity as a function of mimosa cover at Gunbalanya, and Bayliss & Walden unpubl. for the loss of magpie geese nesting habitat in the NT; figs 11 & 12 respectively), for this exercise the control target level was set arbitrarily (1000 ha, see below). This is because the socially (& culturally) acceptable extent of mimosa and corresponding damage level has not been previously defined through a comprehensive consultation process involving all major stakeholders (eg the Gunbalanya community, the NT and Commonwealth governments). The target level is probably the most critical control parameter because it determines the amount of effort required for maintenance control *ad infinitum*, which in turn determines the annual level of investment obligation required to chemically and/or mechanically control mimosa forever, or until a cost-effective biological control program is in place.

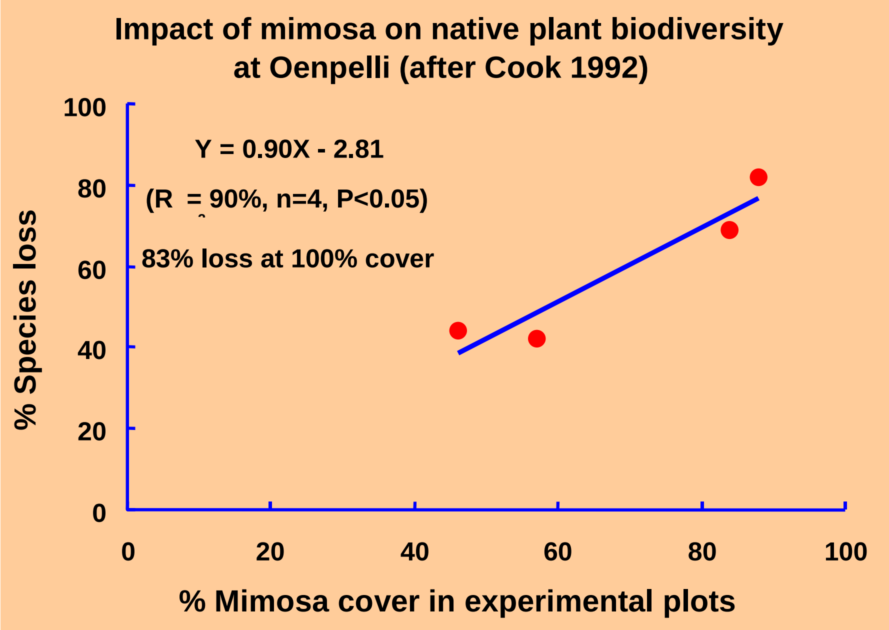


Figure 11 The regression relationship between the biodiversity impact (% of native floodplain species lost) of *Mimosa pigra* and its extent (% cover) in experimental plots on the Gunbalanya floodplains (data from Cook 1992, CSIRO)

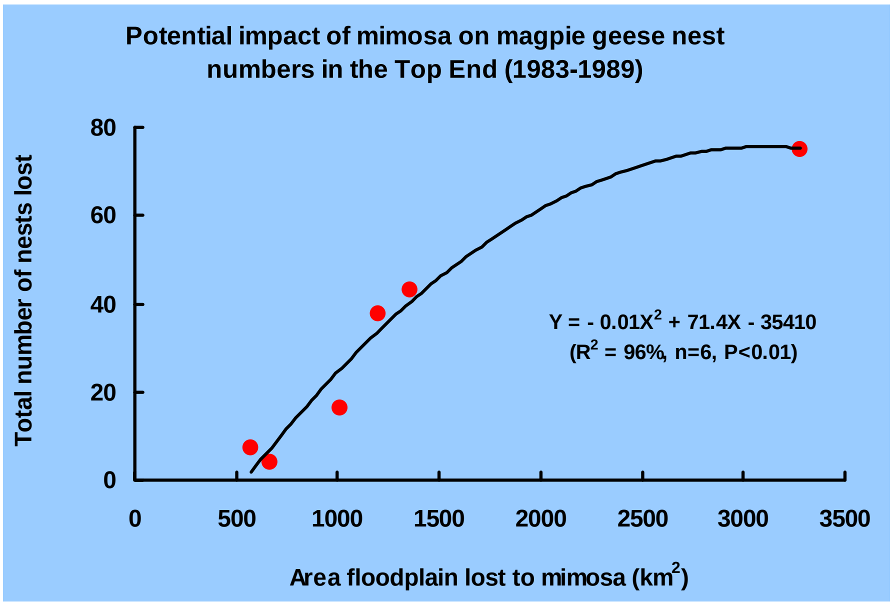
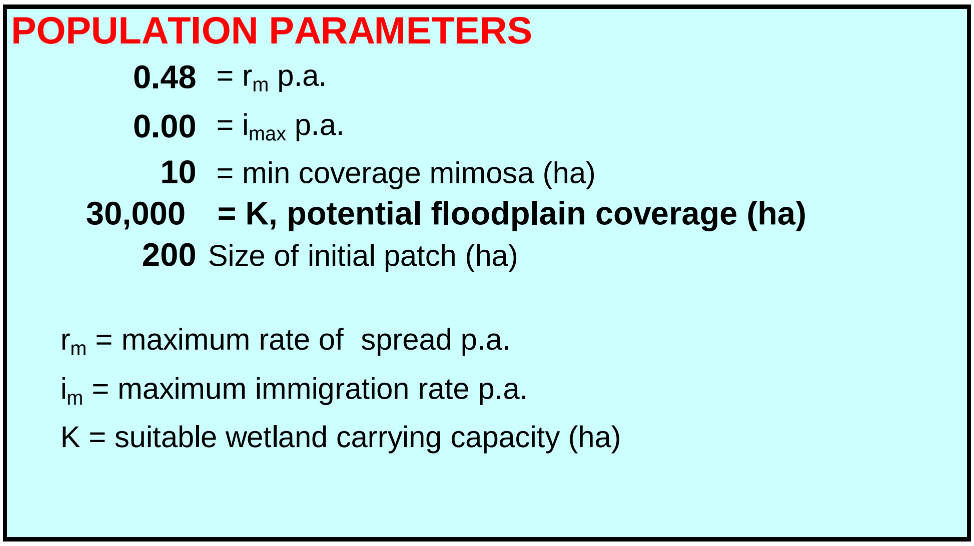


Figure 12 Nonlinear regression relationship between the biodiversity impact (total number of magpie geese nests lost) of *Mimosa pigra* and its extent (area of floodplain to mimosa). Bayliss & Walden (unpubl. data)

The Population Parameters box (figure 13) allows managers to set the parameters for any pest species depending on available knowledge from field or experimental observations, or life history characteristics such as maximum rate of spread and potential carrying capacity (or extent of suitable habitat).



POPULATION PARAMETERS

0.48 = r_m p.a.

0.00 = i_{max} p.a.

10 = min coverage mimosa (ha)

30,000 = K, potential floodplain coverage (ha)

200 Size of initial patch (ha)

r_m = maximum rate of spread p.a.

i_m = maximum immigration rate p.a.

K = suitable wetland carrying capacity (ha)

Figure 13 Population parameters for a simulation of mimosa control on the Gunbalanya floodplain in western Arnhem Land

Figure 14 shows clearly the exponential rate of spread of mimosa observed on the Gunbalanya floodplain between 1980 and 1991 as reported by Lonsdale (1993a), and which was used to estimate maximum spread rate in the above population model (see fig 13, Population Parameters box, r_m ha.y^{-1}). For comparison, the observed and projected exponential spread rates for mimosa on the Adelaide River floodplain are shown also, for the periods 1980 to 1986 and 1987 to 1991, respectively (Lonsdale 1993a). The population dynamics model used in all scenario simulations is an exponential growth model with a ‘ceiling’ defined by the extent of available floodplain. Here all floodplain habitats are assumed to be suitable mimosa habitat and, most likely, approximates reality (Finlayson pers comm.). However, more refined spatial ‘habitat suitability’ and population dynamics models for mimosa and other major floodplain weeds on the Magela will be used in future scenario simulations.

There are three ways to basically manage control operations once a method (&/or combination of methods) has been chosen, and these are to vary: (1) the level of initial reduction; (2) the level of maintenance reduction; and (3) the time interval between reductions. The Model Settings box (figure15) allows managers to choose: the simulation interval; the year and month of treatment; the area treated initially; the area treated during maintenance, and the treatment interval in years. To maintain infestations where impacts are reduced to, or below a socially acceptable level, treatment would need to be equal to, or be greater than, the rate of spread/re-infestation. The Model Settings box (control inputs) has options (not used here) for management scenarios that use a combination of biocontrol (including the ameliorating effects of genetic resistance) and different methods of conventional control (eg a choice between chemical &/or mechanical control). A density floor of 1 ha was arbitrarily used to account for the fact that mimosa cannot be eradicated from extensive areas of floodplain using conventional control methods.

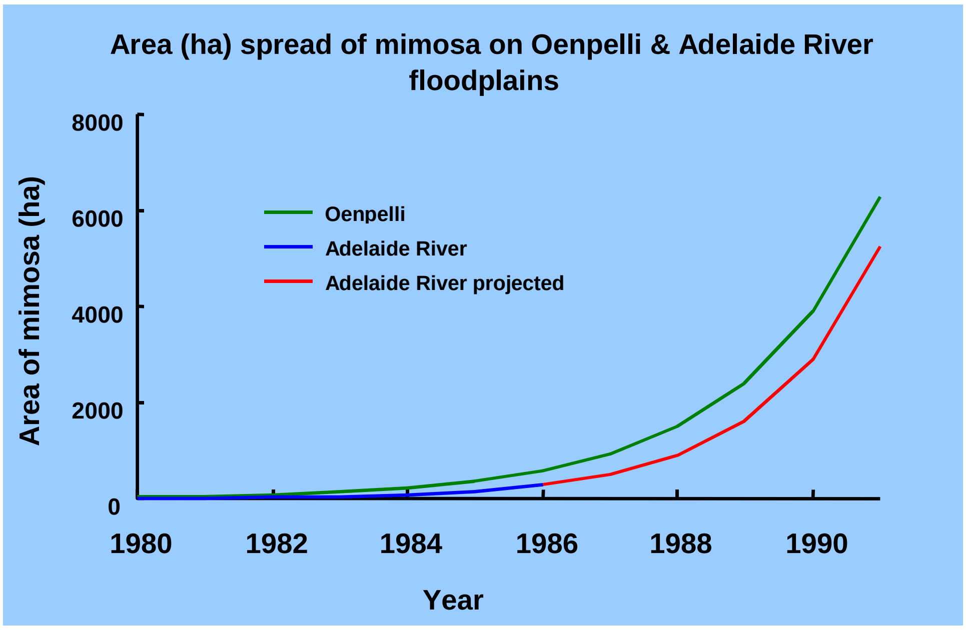
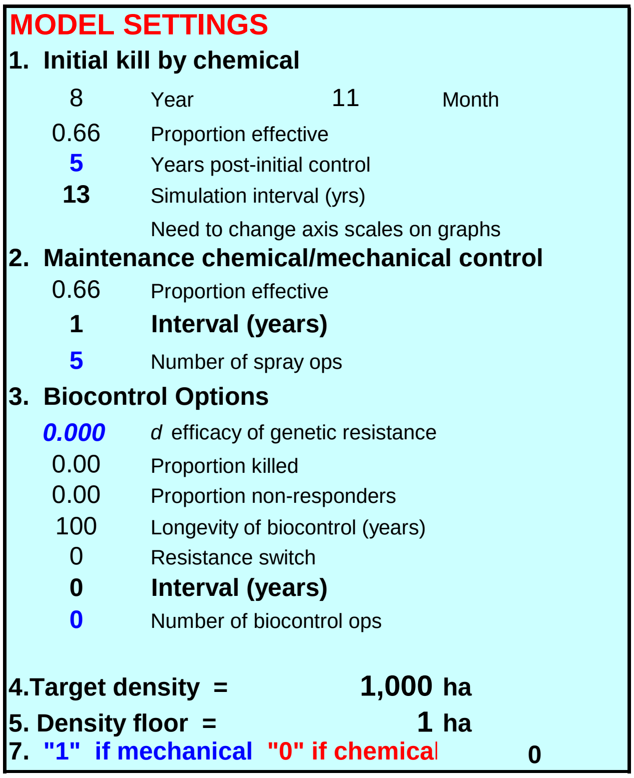


Figure 14 Observed exponential rates of spread of *Mimosa pigra* on the Gunbalunya floodplain (green line, 1980–1991), and that for the Adelaide River floodplain (blue line, 1980–1986). The exponential rate of spread of mimosa between 1987 & 1991 across the Adelaide R floodplain is indicated also (red curve).



MODEL SETTINGS

1. Initial kill by chemical

8 Year 11 Month
0.66 Proportion effective
5 Years post-initial control
13 Simulation interval (yrs)
Need to change axis scales on graphs

2. Maintenance chemical/mechanical control

0.66 Proportion effective
1 **Interval (years)**
5 Number of spray ops

3. Biocontrol Options

0.000 *d* efficacy of genetic resistance
0.00 Proportion killed
0.00 Proportion non-responders
100 Longevity of biocontrol (years)
0 Resistance switch
0 **Interval (years)**
0 Number of biocontrol ops

4.Target density = 1,000 ha
5. Density floor = 1 ha
7. "1" if mechanical "0" if chemical 0

Figure 15 Model settings for a simulation of mimosa control on the Gunbalanya floodplain

Figure 16 tracks the simulated extent of mimosa over time on the Gunbalunya floodplain, before and after control commenced in 1992, using the population dynamics parameters and management settings outlined above. The model suggests that in three years mimosa was first reduced below the arbitrary target level of 1000 ha (10km^2), and that by four years it was kept below the target level. However, the model also shows that should maintenance control cease at the appropriate level, mimosa will increase rapidly past the target level. Anecdotal information suggests strongly that this has happened (Guy McSkimming pers comm.).

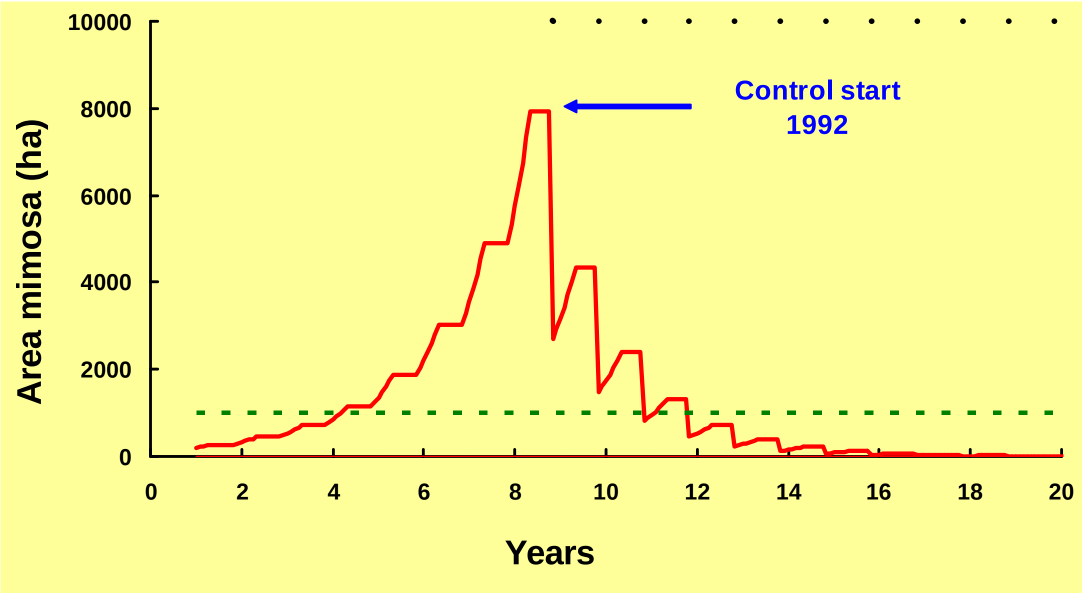


Figure 16 Simulated trends in the extent Mimosa pigra (ha) on the Gunbalanya floodplain before and after control commenced in 1992 (see Model Settings & Population Parameters boxes above): dashed line is the target

The success or efficacy of the control operation can be measured in a number of ways (success indicators), such as: the average extent of mimosa after the initial massive reduction; the number of years to first past the management target; and the number of years mimosa is above and below the management target (see fig 17, Post-control Efficacy box).

	POST-CONTROL EFFICACY
607	Average post-control weed extent (ha)
2	Years to 1st pass management target
7	Years above management target
6	Years below management target

Figure 17 Post-control efficacy indices for a simulation of mimosa control on the Gunbalanya floodplain in western Arnhem Land

Nevertheless, all these indices of control efficacy are pegged to the control target level whereby a socially acceptable reduction in weed damage is obtained. Needless to say, they must also be assessed in relation to the costs-of-control because, taken together, they comprise the benefit-cost analysis used to ultimately determine the success of control.

Figure 18 shows the cost-of-control curve derived for mimosa on the Gunbalanya floodplain using all available cost and effort data between 1992 and 1996. As with the effort – abundance curve for mimosa on Kakadu, the form of the relationship is typically negative exponential; that is, the $ cost per ha to control mimosa by chemical/mechanical means, including survey and monitoring costs, is substantially higher when abundance is lower compared to when abundance is higher.

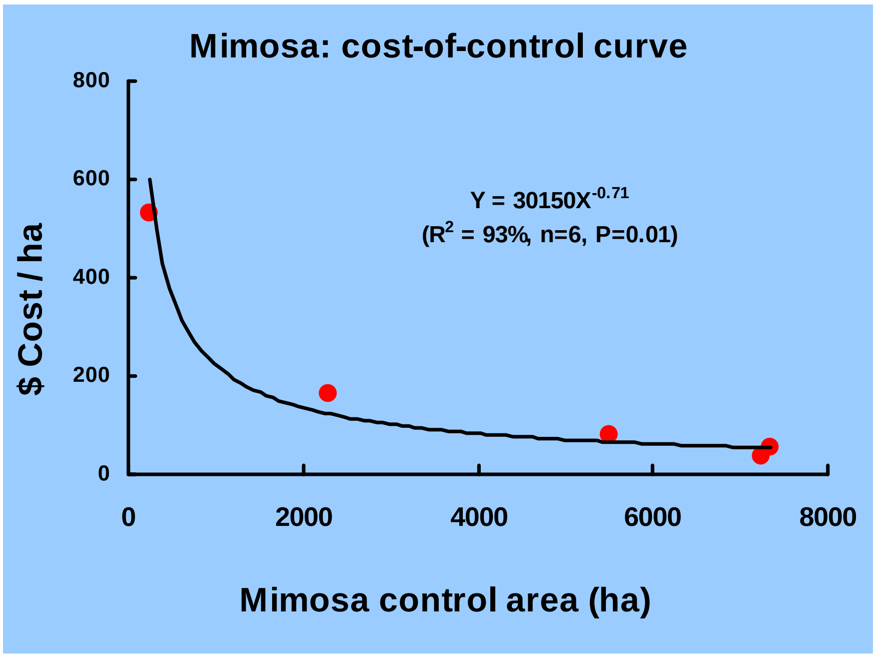
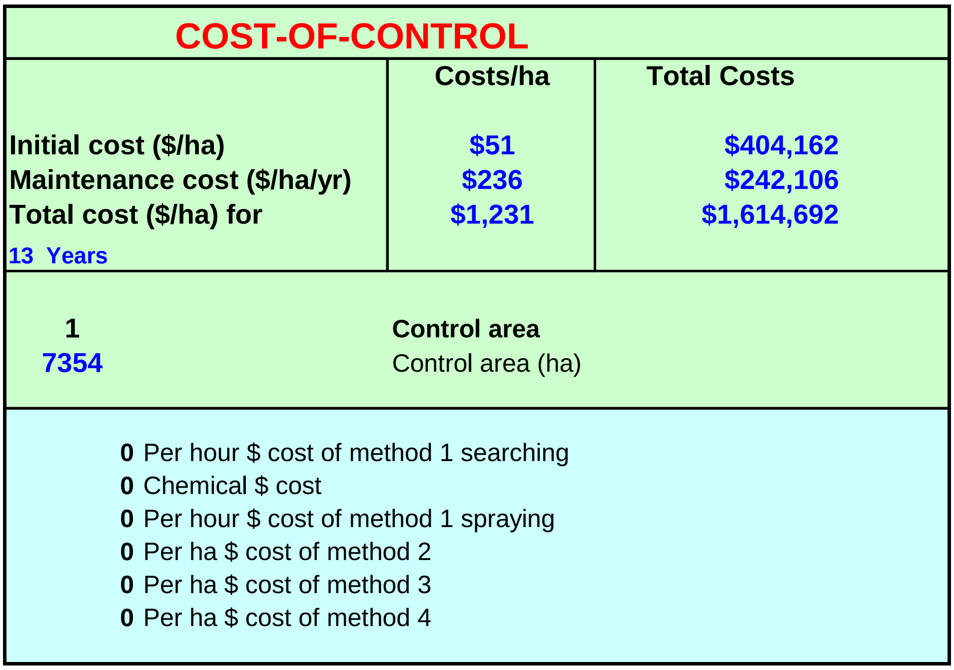


Figure 18 Estimated control cost ($) for large infestations of mimosa on the Gunbalanya floodplain, west Arnhem Land. Operational costs only, and excludes salary and salary related on-costs, and large capital items (eg bulldozers)

For the mimosa control simulation outlined above, Figure 19 (Cost-of-control per Habitat box) summarises the following major cost outputs: initial cost ($/ha); subsequent annual maintenance cost ($/ha/yr); and total control cost ($) for the simulation period (here 13 years). These cost estimates closely approximate the actual operational costs (minus salary & on-costs, & large capital items) reported in the annual budget summaries of the Oenpelli Mimosa Reports prepared by the DIPE (& Miller, pers comm.).



COST-OF-CONTROL

	Costs/ha	Total Costs
Initial cost ($/ha)	**$51**	**$404,162**
Maintenance cost ($/ha/yr)	**$236**	**$242,106**
Total cost ($/ha) for **13 Years**	**$1,231**	**$1,614,692**

1	**Control area**
7354	Control area (ha)

0 Per hour $ cost of method 1 searching
0 Chemical $ cost
0 Per hour $ cost of method 1 spraying
0 Per ha $ cost of method 2
0 Per ha $ cost of method 3
0 Per ha $ cost of method 4

Figure 19 Cost -of -control values for a simulation of mimosa control on the Gunbalanya floodplain. Options for different methods of control are not used here.

5.2 Calotropis

As an aside from the main topic of wetland weeds in the Magela catchment, five years of control data were available from the Mary River District for the weed Rubber bush (*Calotropis procera*). This section demonstrates that the cost-of-control modelling process described above for mimosa, using existing operation data on survey and control effort, can also be effectively applied to other weed species on KNP (see figs 20 & 21). An additional two years of data were extracted from the district's weed diaries, however these records may not have been complete as person days were not always entered and some extrapolation was necessary. As with all weed management programs, this highlights the importance of accurate and detailed records. For example, a few omissions or lack of detail on plot sizes, plant numbers and effort can affect the usefulness of a whole run of years of records.

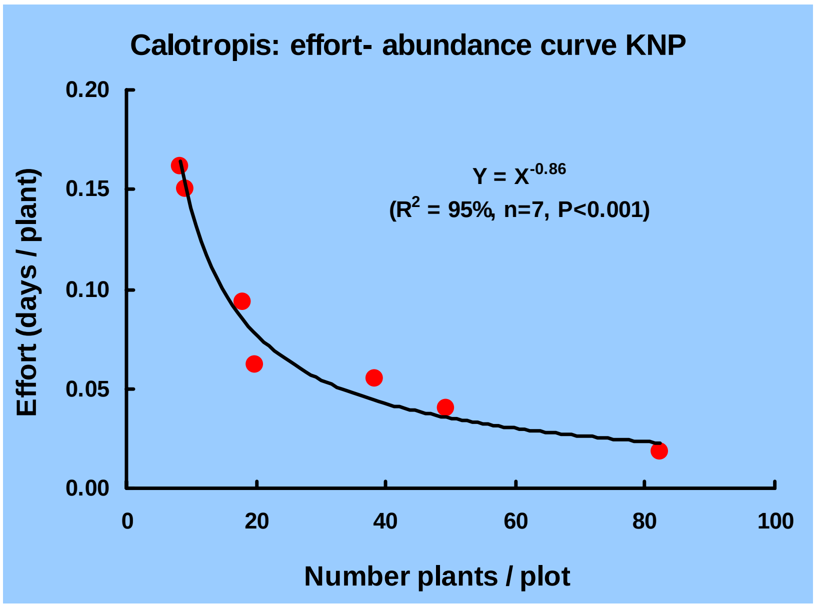


Figure 20 Effort-abundance curve (days per plant vs number plants per plot) for Calotropis in the upper South Alligator Valley

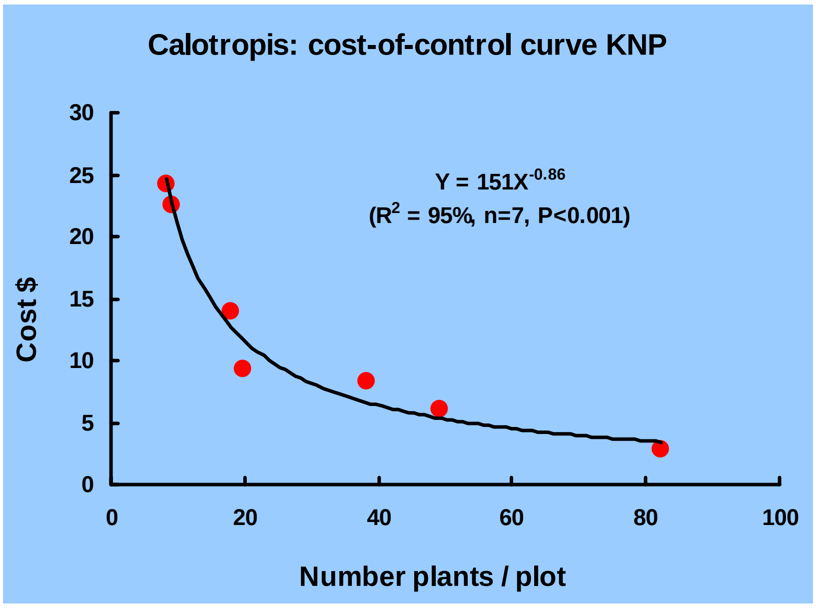


Figure 21 Cost-of-control curve ($ per plot) for Calotropis in the upper South Alligator Valley

6 Additional information

The final risk assessment will contain information on the following topics.

Vulnerable species & habitats

Fauna

Flora

Socio-economic effects

Effects on tourism activities/incomes

Effects on cultural values

Invasion rates and pathways

Preferred habitats and environmental conditions

Potential distribution

(including modelling habitat suitability indices & conducting spatial ecological risk assessments)

Identification of the risks

Land use implications

Mechanisms of seed/propagule transport

Factors affecting colonisation

Consequences of the impacts

Uncertainty and information gaps

Related to the effects

Related to the extent

Management Implications

Education and awareness

Prevention of spread

Research and development

Impact reduction

Monitoring and further research

Conclusions

How these weed species could confound assessment of any mining-related pressures

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