3.9 In situ toxicity monitoring in Magela Creek

Work commenced in the 2005–06 wet season, to evaluate the potential for in situ deployment inside floating cages in Magela Creek of the same snail and fish tests currently used for the creekside monitoring programme. This technique would provide a much more cost effective way of providing almost continuous biological monitoring of water quality in Magela Creek.

Preliminary studies involved developing a suitable design of holding vessels for test organisms, and assessing the responses of freshwater snails (reproduction) to a number of holding conditions and feeding regimes. The 2005–06 wet season testwork also included a comparison between the in situ deployment and standard creekside tests of egg production by snails. This comparison was done near the upstream water quality monitoring site, MCUS (see Map 2, Magela u/s).

While in situ testing has previously been investigated as a technique for biological monitoring in Magela Creek (Annual Research Summary 1987–88, 1988–89, 1989–90 and 1990–91), the method has remained undeveloped until now because of perceived occupational health and safety advantages of the creekside monitoring procedure (in particular, ready accessibility and safety of staff). However, both the high maintenance costs of the existing creekside monitoring coupled with refinement over the years of the protocols for the snail and fish tests have now led to a re-evaluation of the viability of in situ testing.

Apart from low costs, in situ testing has its own inherent advantages over the established creekside monitoring approach. These include removal of reliance on powered pumping systems in an area of high electrical storm activity, improved water flow-through and contact conditions for the test organisms, and portability. These inherent advantages make the method appealing for future monitoring at Ranger and, potentially, also for use at other mine sites in the Northern Territory and elsewhere.

Preliminary in situ tests were run in parallel with the creekside monitoring tests starting on the 17/02/06, 03/03/06 and 07/04/06 near site MCUS upstream of the Ranger mine. These trials investigated one of two possible feeding regimes (i) daily feeding per current creekside monitoring protocol, and (ii) feeding only once, at the start of each four-day test. Daily feeding enabled the direct comparison of results with those from the existing creekside monitoring programme whilst the inclusion of 'feeding once only' (regime ii) enabled investigation of a more cost-efficient feeding regime.

Results from the feeding and in situ versus creekside monitoring comparative tests are presented in Figure 3.11. These data show that snail egg production in the in situ test vessels is comparable to that measured in creekside monitoring tests, and that both feeding regimes should be further evaluated. The results from the daily feeding in situ test are similar to those from the creekside monitoring control site and are almost exactly the same as those from the downstream creekside monitoring site for all three trials (Figure 3.11). The results obtained from the in situ tests in which food was provided only at the start of the test were encouraging, with close resemblance in egg production to that found for the daily feeding in situ test in the first two trials. If start-only feeding can be used for the in situ method, this will have substantial benefits for staff resourcing. It will also mean that this monitoring technique will be much more viable for extended deployment at less accessible (for example, Gulungul Creek) or more remote locations.

An established baseline of creekside monitoring test data has been obtained since 1991–92 (Figure 2.8). Hence it is very important to ensure that the new method yields comparable results before it can be phased in as the sole procedure in the future. To this end, further testing of the in situ vessels and feeding regimes will occur over at least two wet seasons, conducted in parallel with creekside monitoring and extending also to the downstream site. Future comparative tests will focus on the paired-site monitoring design employed for creekside monitoring (described in Chapter 2 of this Annual Report) and compare the 'differences' in responses between upstream and downstream sites for both test conditions and feeding regimes.

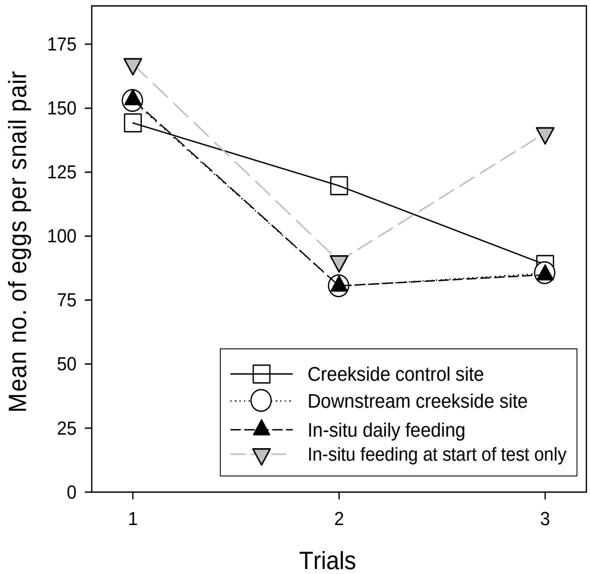


Figure 3.11 Comparison of freshwater snail egg production for routine creekside monitoring and two feeding regimes of in situ toxicity monitoring

3.10 Ecological risk assessment of Magela floodplain from diffuse landscape-scale threats and point source mining threats

This ecological risk assessment is the final project of the landscape-scale analysis of impacts programme, established in 2002 to help differentiate mining and non-mining impacts on the Ramsar-listed Magela Creek floodplain wetlands downstream of the Ranger uranium mine. Ecological risk assessment allows the level of risk to the 'health' of ecosystems exposed to multiple stressors to be quantified in a coherent, robust and transparent manner. A high protection level for the biodiversity of aquatic ecosystems was used as the assessment endpoint, so conclusions here can be regarded as being appropriately conservative.

The key findings from four projects are presented in two parts. Part 1 reports results from a first-cut quantitative ecological risk assessment of the threats posed by diffuse non-mining landscape-scale factors to the condition of selected World Heritage values on the Magela floodplain. Part 2 similarly reports an ecological risk assessment of four key chemical contaminants released from Ranger into the surface water pathway of Magela Creek as mapped out by the conceptual contaminants pathways model presented in the Supervising Scientist's 2004–05 Annual Report.

3.10.1 Part 1: Landscape

The status of past and current ecological values (assets) and threats on Magela floodplain were mapped in a GIS to facilitate spatially explicit risk assessment. Shape-file data layers were converted to raster grid-cell data format at appropriate levels of spatial resolution (here 250 m x 250 m cells). The spatial and temporal scope of many data sets extended beyond the Magela catchment to include Kakadu National Park and the Alligator Rivers Region in general.

Choice of World Heritage values

The two key 'susceptible' World Heritage (and Ramsar) values chosen for assessment are waterbirds and native wetland vegetation. An assessment of change in their 'condition' was undertaken using spatial and temporal data obtained between 1981 and 2003. The initial focus has been on changes in the abundance of 'indicator' species of plants and waterbirds. Future work will examine possible changes in biodiversity.

For vegetation the focus was on: weeds; important bush foods of traditional Aboriginal owners (for example, the red lily); key habitat components of waterbirds for nesting and food (wild rice, *Oryza meridionalis* and *Eleocharis* sedges); and riparian trees susceptible to saltwater intrusion and/or fire (Melaleucas).

For waterbirds, the iconic magpie goose and egret were chosen for initial analysis because they are the most seasonally abundant plant and fish eating birds, respectively, that forage on the Magela floodplain. Additionally, the magpie goose is an important part of the diet of traditional Aboriginal owners in the Alligator Rivers Region.

Results from the long-term observational record of changes in billabong fish community structure were reported in section 3.6 of the Supervising Scientist's 2004–05 Annual Report. Between 1989 and 2005 (16 years) the chequered rainbow fish and two species of glassfish at the Mudginberri monitoring site downstream of Ranger exhibited long-term declines in abundance (13% per annum on average) that are apparently unrelated to mining impacts. Three key correlates and associated working hypotheses that could explain these declines are: (i) increases in mean wet season flow leading to lower water solute concentrations known to be harmful to larval rainbow fish; (ii) decreases in the period of annual drying of the floodplain potentially leading to reduced release of nutrients upon floodplain re-wetting (flood-pulse theory), and thereby reducing fish production in this important breeding and recruitment zone; and (iii) increases in the extent of floodplain grasses, including the introduced para grass, thereby reducing habitat availability and pathways for upstream migration of fish recruits.

Choice of landscape-scale threats

Four major categories of landscape-scale threats to the above selected World Heritage values were identified:

- invasive species – specifically the wetland weeds mimosa, para grass and salvinia, and feral pigs (classified as a 'Threatening Process' under the *Environment Protection and Biodiversity Conservation Act 1999*);
- unmanaged fire;
- infrastructure (eg towns and mines); and
- potential climate change impacts (rising sea level and concomitant salinisation of freshwater ecosystems, increasing intensity and frequency of storms).

Medium to long-term climate change threats, although highly relevant, were beyond the scope of this study.

The ability of wetland weeds to dominate and completely alter aquatic ecosystems has been well documented. The Magela floodplain fortunately remains free of mimosa because of a highly successful 'search and destroy' programme by National Park rangers. The impact of the floating fern salvinia has been greatly reduced by biological control. Hence, para grass was the primary focus of our risk assessment for weeds.

Ground disturbance damage caused by feral pigs has been ascertained across Kakadu during systematic aerial surveys in 1985, 1995 and 2003, and its damage to natural and cultural values has been documented in consultancy reports to Kakadu National Park management.

Dry season fire on the Magela floodplain can be viewed either as a cultural asset if part of an indigenous burning regime, or as a threat if unprescribed by traditional Aboriginal owners or Park management. To determine whether or not fire on the floodplain should be viewed as an asset or a threat would require detailed ethno-ecological knowledge beyond the scope of the present study. However, such a study was completed in the adjacent South Alligator River catchment at Boggy Plain wetland as part of the landscape programme. Infrastructure in the vicinity of the Magela floodplain comprises mostly roads, tracks and fence lines, and these may facilitate the spread of weeds and possibly cause erosion and siltation. Additionally, the minesite and associated township are sources of weeds for the Magela catchment and Kakadu in general.

Key results

Vegetation

The following eight classes of native vegetation were used to analyse change between 1983 and 2003: *Eleocharis* spp, *Oryza* spp, *Pseudoraphis spinescens*, *Hymenachne acutigluma*, *Melaleuca* spp, *Nelumbo nucifera*, *Nymphoides* spp and *Leersia hexandra*. Relative change in abundance was measured by change in percentage cover. The following changes were observed: *Nymphoides* and *Leersia* were not recorded in 1983; *Eleocharis*, an important dry season food of magpie geese, decreased by 57%; Melaleucas decreased by 10%; and *Nelumbo* decreased by 85%. The 10% relative change in paperbark forest and woodland is significant because, on an absolute basis, this corresponds to 5 km^2 or 3% of the floodplain.

Mimosa has been kept under control since the early 1980s through an annual investment (and in perpetuity) of approximately $0.5 million by Kakadu management, and para grass and salvinia have since colonised the floodplain (see weeds section below).

Waterbirds

The number of magpie geese that occupy Magela floodplain in the late dry season has decreased on average by 7% pa between 1981 and 2003 (22 years) (Figure 3.12a, dashed line). In the wet seasons of the early 1980s, magpie geese used the central portion of Magela floodplain for nesting, an area now occupied extensively by para grass (see 2005 map in Figure 3.15). Additionally, in contrast to the early 1980s, areas now colonised by para grass are used less extensively for feeding by magpie geese in the late dry season, and this is possibly related to the 57% reduction in the extent of *Eleocharis* sedge. Similarly, fish eating egrets decreased on average by 9% pa since 1981 and by 2003 they also altered their dry season distribution.

Although the floodplain is about 200 km^2 in area it is still too small to be able to place into context long-term changes in the abundance of highly mobile waterbird species in isolation from regional and national trends. In particular, the effects of anthropogenic-induced changes (for example, the commencement of mining at Ranger; the reduction in buffalo numbers and concomitant increase in pig numbers; and the colonisation of the floodplain by para grass weed; see below). Fortunately concurrent regional and national survey data for magpie geese are available, including surveys started in 1958 by CSIRO and continued to the present by the NT Parks and Wildlife Commission. Analyses at increasing spatial scales (Figure 3.12b) show that trends in the observed abundance of magpie geese in the late dry season on the Magela floodplain were highly concordant with similar trends for the same time period across the Alligator Rivers Region (Figure 3.12a, solid line & filled circles), and between 1983 and 1996 for the 'Top End' of the Northern Territory (Figure 3.12a, solid line & open circles). The latter area includes most of the Australian goose population.

The Northern Territory surveys provide 45 years of almost continuous data that suggest cycles of magpie geese numbers over decadal time scales (20–25 year periods; trendline in Figure 3.12c and verified by time series analysis). Early studies show that the population dynamics of magpie geese are driven to a large extent by deviations in mean annual local rainfall in river catchments, which itself exhibit similar decadal cycles.

The driving variable of magpie geese population dynamics is wetland vegetation (ie, for food and nesting material), which is highly correlated to flow, water depth and ultimately rainfall. Time series and CSUM (cumulative sum of the mean deviations) analyses of wet season flow at three long-term gauging stations across the Northern Territory (Magela Creek at G8210009, 1972–2005; Daly River, 1961–2005; and Katherine River, 1958–2005) all show similar and concordant 20-year periods as that for magpie geese numbers, and are highly cross-correlated. Similar results have been found between flow and indices of barramundi abundance (catch-effort) in the Daly River (via the Tropical Rivers Inventory Assessment Project, Section 3.11).

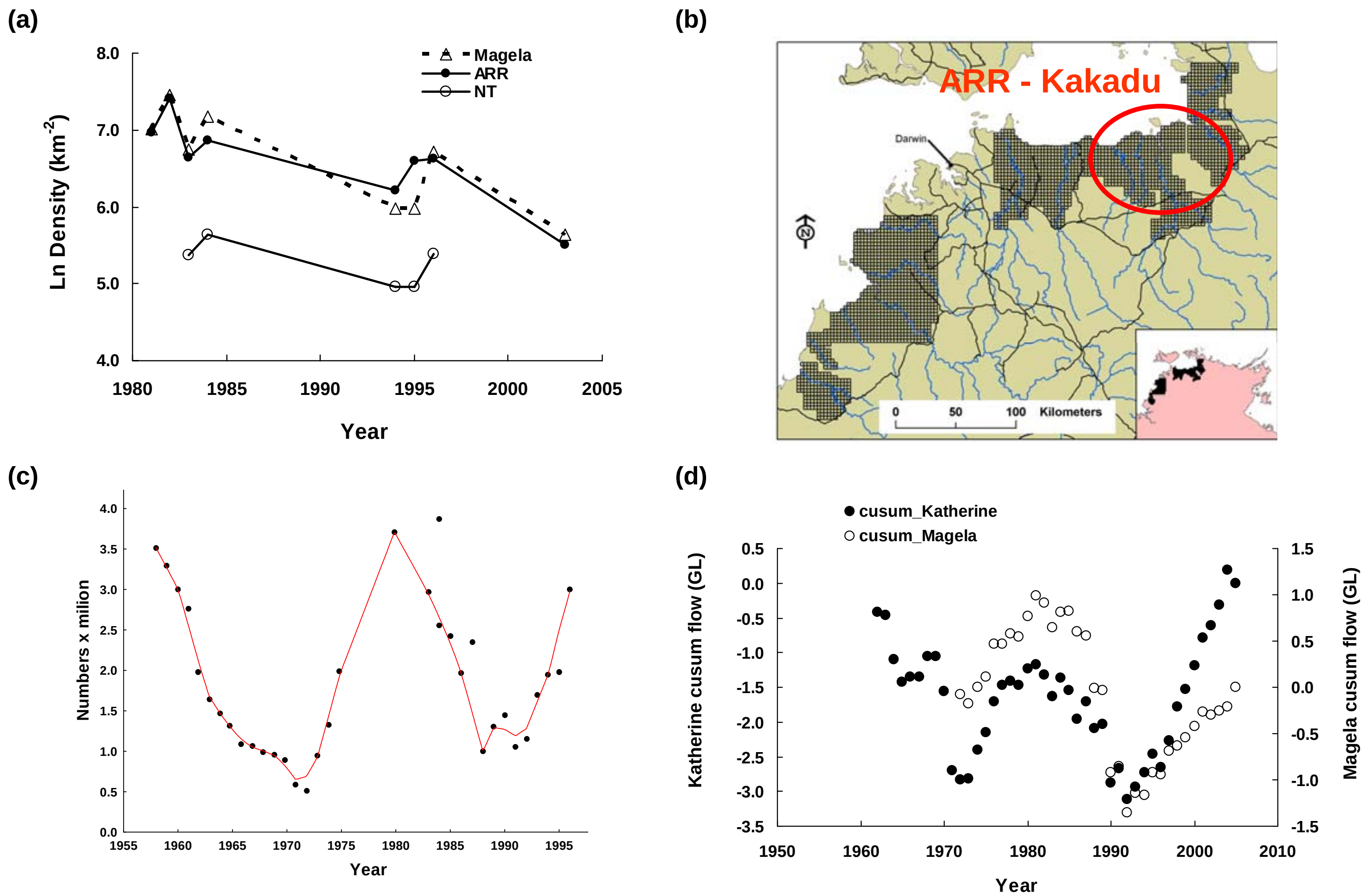


Figure 3.12a–d. (a) Concordant trends in dry season density of magpie geese for the Magela and the Alligator Rivers region (ARR)(1981–2003), and the 'Top End' of the NT (1983–1996). (b) Survey area (shaded) of the annual magpie geese surveys in the NT. Circle is the ARR encompassing the Magela floodplain (K Saalfeld, NT P&WC). (c) Estimated numbers of Magpie Geese in the NT between 1958 and 1996, with a Lowess smoothed trend showing 20-year periods. (d) Cumulative sum (CSUM) of the mean deviations in wet season flow (GL) for Magela Creek at G8210009 (1972–2005) and Katherine River (1958–2005), also showing 20-year periods.

Hence, because of the possibility of large-scale ‘external’ ecological drivers, any meaningful assessment of World Heritage values of waterbirds and other highly mobile water-dependent ‘indicator’ species on Magela floodplain needs to focus on the condition of their seasonal in situ habitats and not trends in abundance.

The relatively long-term patterns of decline in magpie geese, egrets, rainbow and glassfish on Magela floodplain are all most likely related to decadal flow patterns, although local anthropogenic causes cannot be ruled out (for example, the loss of key wetland habitat due to para grass colonisation).

Weeds

In the early 1980s, para grass was present in very small areas of the Magela Creek floodplain. In the mid 1990s, para grass in the vicinity of the largest infestation spread from 132 ha to 422 ha in five years (1991–1996). This core patch of para grass occupies the centre of the floodplain (see Figures 3.14 and 3.15) and is expanding on average at 14% p.a. (Figure 3.13a). That is, doubling in extent every five years. The increase in area of para grass between 1991 and 1996 showed a corresponding decrease in area of a community of wild rice and *Eleocharis* sedge (this study), important food resources for pre-fledging and adult magpie geese, respectively. Para grass currently occupies about 1250 ha (or 10% of the floodplain with 100% cover), with new outbreaks occurring in inaccessible dense Melaleuca woodland.

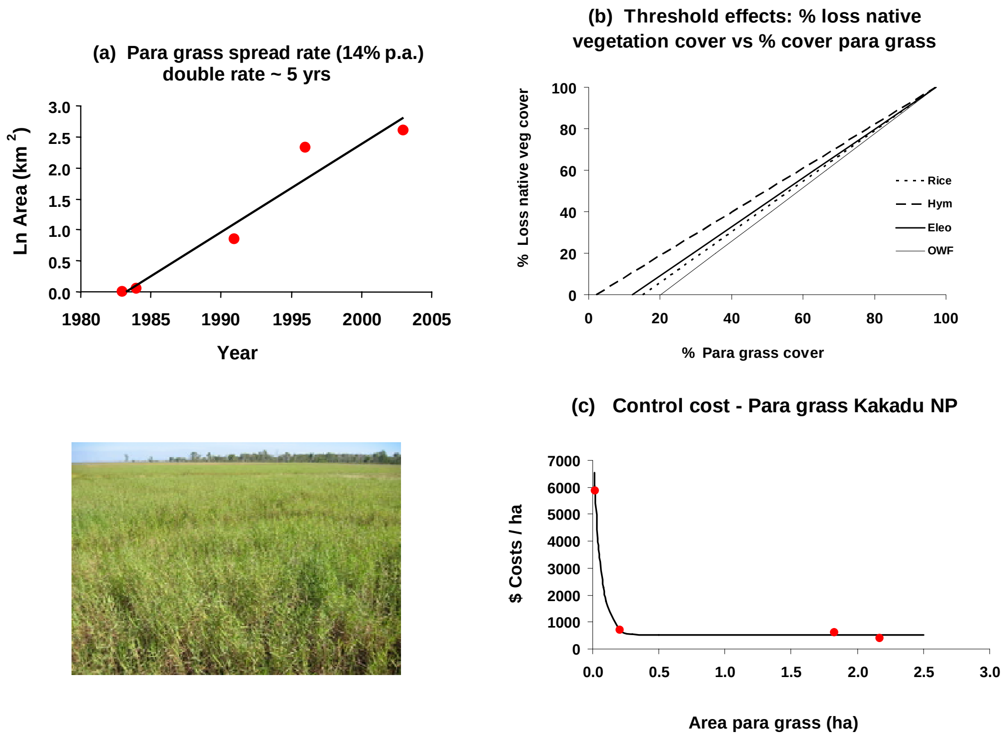


Figure 3.13a–c. (a) Linear regression between $\mathrm{Log_e}$ extent (km^2) of para grass and time (years) (R^2 = 69%, P<0.04). (b) Relationship between loss of native vegetation cover (%) of four key wetland plants and increasing cover of para grass. (c) Control-cost curve for para grass (Noulangie floodplain, Kakadu).

Data obtained from sample plots in 2003 show that the percentage of native vegetation (for example, wild rice, *Eleocharis*, *Hymenachne*, open water/lilies & *Leersia*) 'lost' to para grass rapidly increased with increasing weed cover, and importantly that there was a 'threshold' effect for each plant group (Figure 3.13b, minus *Leersia*). Hence, for most floodplain plants, measurable impacts did not occur until para grass reached 15–20% cover, suggesting that this extent of cover may represent a pragmatic, cost-effective and justifiable control target.

Cost-of-control functions have been developed by ***eriss*** for mimosa and now para grass (Figure 3.13c), and are critical for evaluating the benefits and costs of any invasive species management programme, which are essentially risk management programs. The cost-curve for para grass shows that a 15–20% control target would avoid exponentially increasing control costs generally associated with unachievable eradication objectives, or cost-prohibitive 'trace level' objectives (this reasoning may not apply to mimosa, however, because of its massive seed set).

A Bayesian Habitat Suitability Model (HSM) was developed in collaboration with Charles Darwin University to predict current and future distribution (exposure) of para grass and, hence, potential impacts on native wetland vegetation. The methodology has been successfully applied to the Mary River floodplain. The risk-based exposure map incorporates test data from high resolution Quickbird satellite imagery (validated by helicopter and airboat surveys) to provide more reliable information on para grass extent over different temporal and spatial scales (Figure 3.14). The methodology developed to date in the core para grass area of central Magela provides a valuable cost-effective monitoring and assessment tool for park managers.

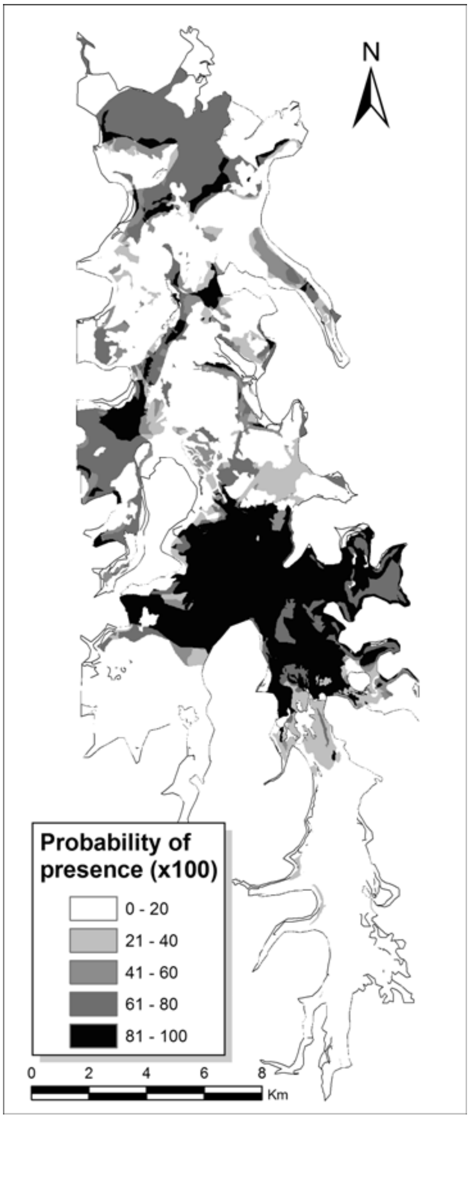


Figure 3.14 Bayesian habitat suitability model for para grass showing exposure probabilities (black is 100% exposure risk or present, white is no exposure risk or absent, with grey scales represening exposure risk in between). The exposure risk map was derived from a Quickbird satellite data capture, helicopter and airboat validation surveys, and GPS observations by Park staff.

A spread rate model was developed to predict para grass extent and, hence, potential ecological impacts over time. Management scenario simulations were undertaken ranging from 'do nothing' to a range of initial and maintenance control investments. Initial simulation results suggest that with no control 50% or more of the floodplain will be lost within 20 years (Figure 3.15). However, this time frame may be the 'best case scenario' because satellite patches of para grass are now spreading along the entire length of the Magela floodplain, representing new centres for expansion.

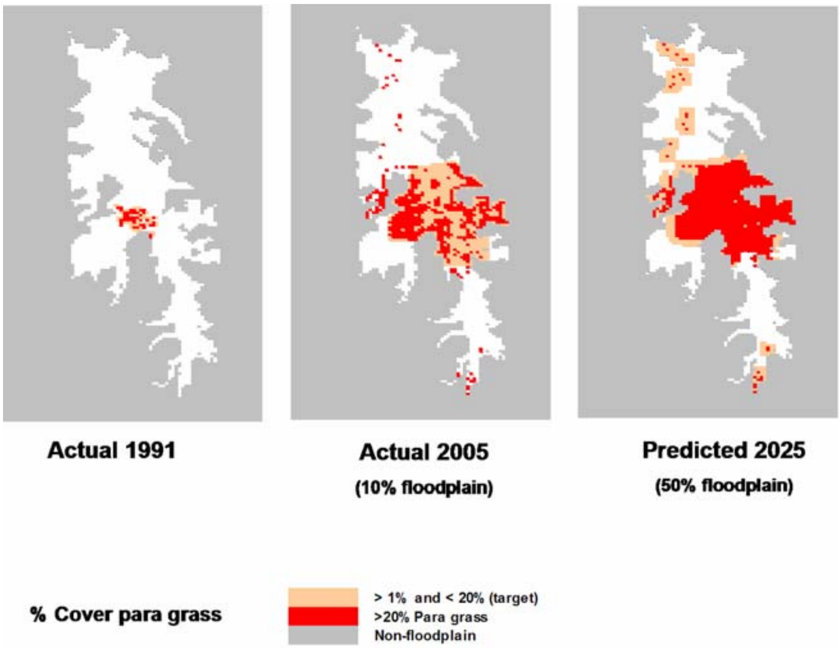


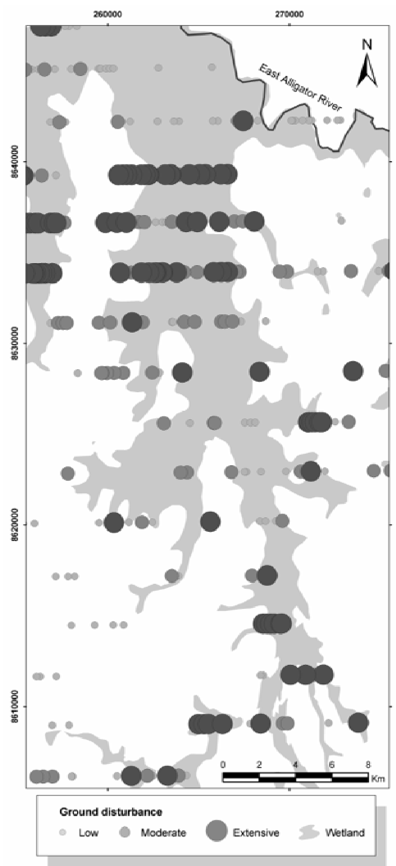
Figure 3.15 The extent of para grass on Magela floodplain in 1991 and 2005, and the predicted extent in 2025 based on habitat suitability, known spread rates and location of known infestations

Pigs

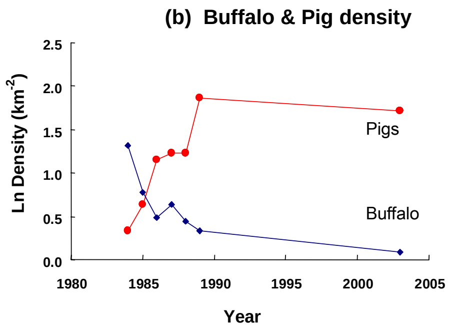
Data for pig density and damage across all floodplains in the Alligator Rivers Region has been used to infer the situation on the Magela floodplain. The extent of pig damage was recorded in four classes (zero, low, intermediate and high) along aerial survey transects divided into 2 km or 5 km units depending on transect spacing. Frequency of occurrence data was then used to estimate probabilities of overall damage.

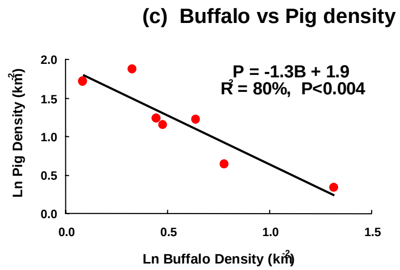
No damage was recorded in 1985, corresponding to low pig densities during the early 1980s when buffalo densities were high.

In contrast, extensive pig damage now occurs across the whole of Kakadu National Park, particularly on floodplains such as Magela (Figure 3.16a). These changes corresponded to a rapid increase in pig numbers following sustained commercial harvesting of buffalo in the 1970s and the Brucellosis and Tuberculosis Eradication Campaign (BTEC) between 1985 and 1989. During this period pig densities increased at a rate close to their maximum rate of population increase for the region (27% per annum, Figure 3.16b).



(a) 2003





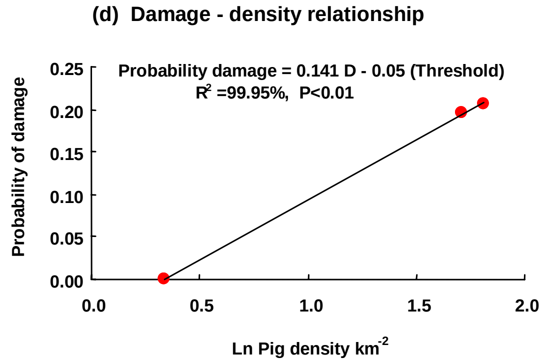


Figure 3.16a–d (a) Distribution and intensity (low, medium & high) of ground disturbance damage caused by feral pigs on Magela floodplain (Nov 2003). (b) Trends in buffalo and pig density (1985–2003) in the ARR (c) Inverse relationship between density of buffalo and pigs in the ARR. (d) Threshold relationship between probability of pig damage and Ln density.

There is a negative relationship between the density of buffalo and pigs since 1983 (Figure 3.16c). A pig control programme was started in 1989 in the northern section of the Park, reducing numbers to about 20% of the initial count by 2003. This probably represents a 'sustained-yield' because the cull rate does not exceed the observed maximum rate of population increase.

Although ground disturbance damage caused by pigs has only been systematically recorded in three of a dozen aerial surveys across the Alligator Rivers Region since 1985, damage and density are nevertheless correlated in a predictable manner and the relationship exhibits a threshold for effects (Fig 3.16d). Control cost functions have been developed by ***eriss*** for pigs in Kakadu using control data collected by Kakadu rangers; the same shoot data were used to derive estimates of absolute density.

Fire

The Bushfires Council of the Northern Territory provided 25 years of Landsat based fire-scar maps for the Northern Territory (1980–2004), which were used to estimate fire occurrence on the Magela floodplain and surrounding Eucalypt woodland.

The frequency of fire over 25 years in each 250 x 250 m grid cell was converted to probabilities for risk assessment. Previous studies have shown that, if the risk of fire in any given location on the floodplain was greater than 0.25 (one in four years), it can be considered a threat to biodiversity values because of suppressed abundances of fire sensitive plant species. However, until recently little was known about the impacts of dry season fires (frequency, intensity and duration) on the structure and composition of wetland vegetation on the Magela floodplain.

Results from the Boggy Plain fire study show that the diversity of wetland plant species increased with the extent of traditional burns if floodplain vegetation was dominated by monocultures of grasses such as *Hymenachne*, particularly if time since last burnt was greater than five years. Therefore, in the ecological risk assessment for fire uncertainty was incorporated by setting the effects probability to 0.50 (that is, hedging bets either way).

The floodplain and surrounding woodland have markedly different fire risk profiles (see histograms in Figure 3.17). Fire on the floodplain occurs on average once every five years (mean P=0.20, median =0.13). In contrast, the surrounding Eucalypt woodland burns on average once every two years (mean P=0.52, median 0.53). A comparison of the shapes of their probability density functions (based on relative frequency histograms) show that fire risk is greater in the woodland because it is more uniformly spread across the entire probability range.

Landscape-scale ecological risk assessment

For landscape-scale (and mining) threats risk is quantified as the probability of an adverse event, or the likelihood of exposure multiplied by the consequences or effects of that exposure (that is, probable risk equals probable exposure times probable effects). Bayes's Theorem was used to derive individual and combined ecological risks. Probability density functions (pdfs) fitted to exposure and effects frequency data were used with Monte Carlo simulation to account for uncertainty.

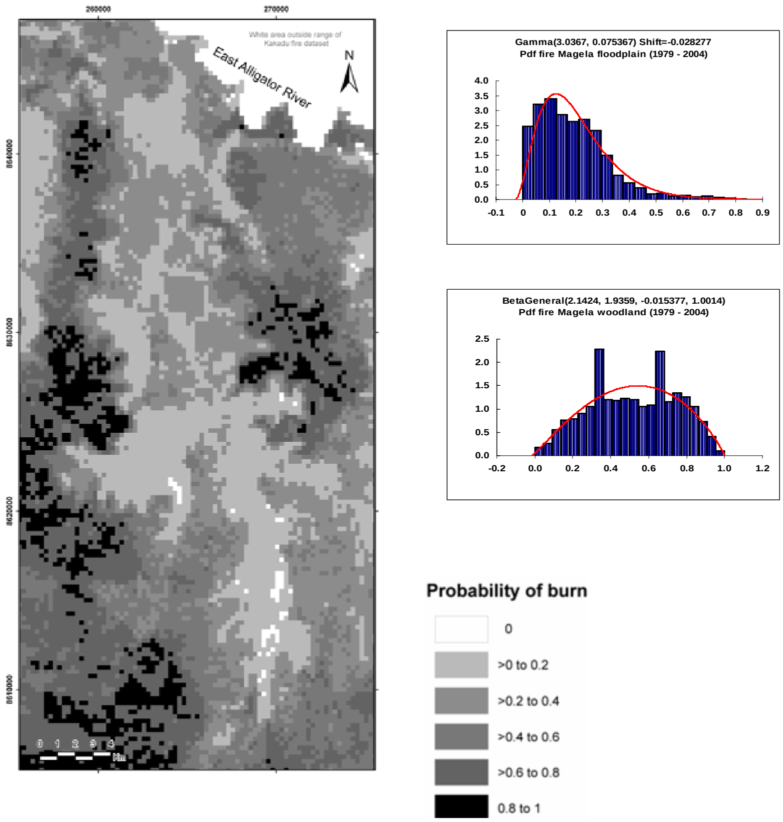


Figure 3.17 Risk of fire on the Magela floodplain and surrounding woodland (1980–2004). Histograms are probability density functions (pdfs) for frequency of fire occurrence on Magela floodplain (top) and surrounding woodland (bottom).

Sensitivity analysis (via regression/correlation methods) was used to rank negative and positive contributions of all inputs into the risk assessment and plotted as Tornado graphs. The end product of the Monte Carlo simulations is a probability density function that characterises all landscape-scale ecological risks in combination, and which can be compared to a similarly derived quantitative risk profile for potential mine-related impacts (see below).

The analysis of the data followed the sequence:

(a) for each risk group derive a probability density function (pdf) for exposure probabilities (P_{exp}) based on spatially derived frequency data in each 250 x 250 m cell across the floodplain (only para grass and salvinia are assessed here as mimosa exposure is < 1%)

(b) derive a pdf for effects probabilities (P_{eff}) of each risk group based on expert knowledge, the literature, and/or experimental or empirical data (if completely unknown set the uncertainty level to 0.50 as for fire; for all others adopt the precautionary principle and set the effects probabilities to 1.0, hence risk assessments will be weighted towards exposure); for each group derive ecological risk ($P_{risk} = P_{exp} \times P_{eff}$) profiles from the exposure and effects probability density functions above using Monte Carlo simulation (10 000 simulations);

(c) combine the risk profiles of all groups;

(d) use sensitivity analysis to rank the contribution of each risk group to total risk, and ascertain the dependencies between and within groups to total risk.

Initial results are presented below. Examples of pdfs for exposure threats are provided above for the occurrence of fire on the Magela floodplain and surrounding woodland (Figure 3.17). The mean landscape ecological risk is 0.21 (Figure 3.18a), with para grass ranking first, pig damage second and uncontrolled fire third (Figure 3.18b).

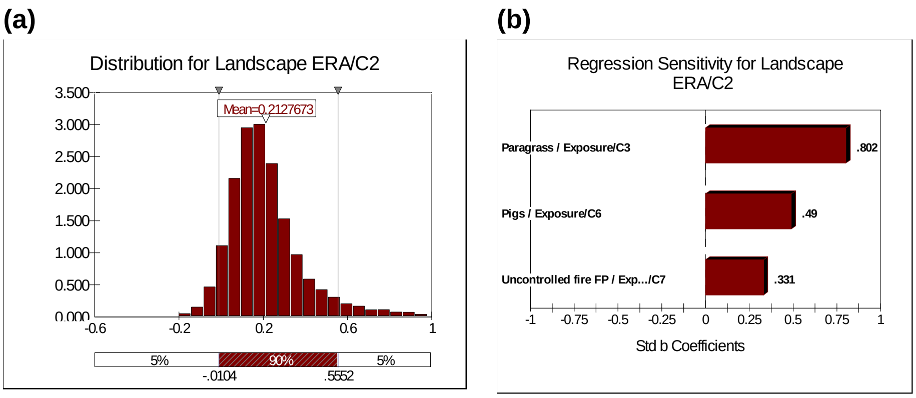


Figure 3.18a & b. (a) Distribution of combined ecological risks for landscape threats. (b) a Tornado diagram illustrating the relative contributions of each risk group to the combined ecological risk assessment for landscape threats

3.10.2 Part 2: Ranger – surface water pathway of chemical contaminants

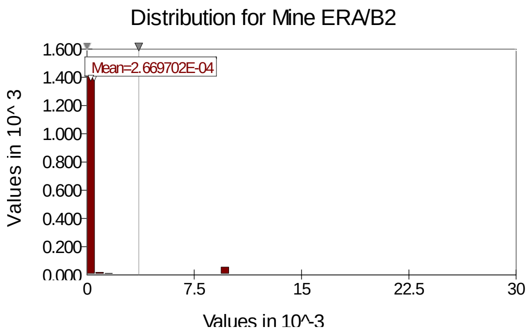
A similar ecological risk assessment process was used to assess four key solutes found in surface water and seepage discharged from Ranger uranium mine into the surface waters of Magela Creek during the wet seasons between 1998 and 2005. These solutes are uranium, magnesium, sulfate (SO_4) and manganese. Weekly point sample data at the downstream statutory monitoring site (Map 2, Magela d/s) were used to characterise off-lease exposure probabilities to aquatic ecosystems downstream of Ranger on Kakadu National Park.

Ecotoxicological end points for uranium and magnesium were used to derive Species Sensitivity Distribution models in order to predict the contaminant concentration (trigger values) that protect 99% of susceptible aquatic species. The models contain a small yet strategic sample across trophic levels and, comprise the most robust quantification of ecological effects by any single hazard to date. The magnesium effects model is complex because mine-derived calcium ameliorates the toxicity of magnesium. However, recent ecotoxicological work shows that a magnesium:calcium ratio of 9:1 is the threshold for magnesium effects and, hence, this value is incorporated into the risk modelling. The trigger value of 1200 ug/L for mangenese recommended by the National Water Quality Guidelines (NWQG) is based on ecotoxicological end points and so is adopted as an interim value for 99% species protection. A 'low reliability' trigger value of 15 mg/L for sulfate was adopted based on limited site-specific effects data (van Dam pers. comm.).

Best-fit exposure probability density functions were produced for each of the four solutes described above.). The details of this process will also be reported later, and initial results are presented below.

In contrast to the normal-like distribution of the combined landscape risk profile (Figure 3.18a), the combined pdf for minesite risks is approximately bimodal with 90% of values clustered closed to zero (Figure 3.19a). The mean minesite ecological risk of one simulation with 10 000 iterations was only 2.7 x 10^{-4}. Uranium exposure made an extremely small contribution. Similarly, the three other solutes posed insignificant risks because exposure probabilities never exceeded the 1% species-affected concentrations, or other relevant effects thresholds (Figure 3.19b).

(a)



(b)

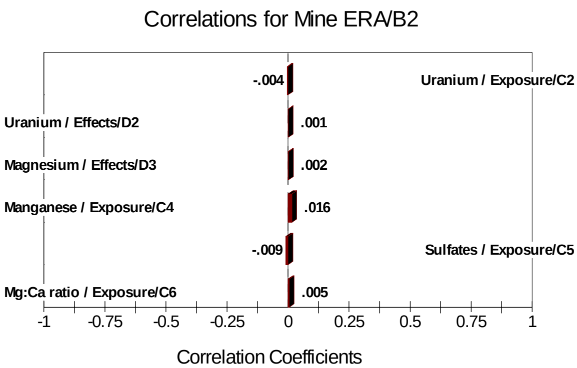


Figure 3.19a&b (a) Distribution of combined ecological risks for minesite chemical threats (note X-axis values are by 10^{-3}), and (b) a Tornado diagram illustrating the relative contributions of each chemical to the combined ecological risk in the surface water pathway.

3.10.3 Conclusions

Two key results from the landscape ecological risk assessment are:

(1) non-mining landscape-scale risks are currently several orders of magnitude greater than mining risks (Table 3.3), although that difference may reduce when on-site water management systems at Ranger change in the transition between mine production and mine closure and rehabilitation; and

(2) Para grass weed (*Urochloa mutica*) is currently the major ecological risk because of its extent (10% cover), effect (a monoculture that displaces native vegetation and wildlife habitat) and rapid spread rate (14% per annum).

The risk posed by para grass has been examined in greater detail by combining the Bayesian habitat suitability model with a spread rate model, therefore encompassing current and future risk to floodplain habitat diversity depending on distance to source and invasion pathways. The overall findings from this landscape ecological risk assessment strongly imply that non-mining landscape-scale risks to Magela floodplain should now receive the same level of scrutiny as applied to uranium mining risks, including an assessment of what appropriate level of investment would be needed to manage these risks.

TABLE 3.3 COMPARISON OF LANDSCAPE AND MINESITE ECOLOGICAL RISKS TO MAGELA FLOODPLAIN, AND THEIR RELATIVE IMPORTANCE RANK

Category	Pathway	Hazard	Risk rank	Action	Time frame
LANDSCAPE	Park-wide	Para grass weed	1	Take active control	In perpetuity
	Park-wide	Pig damage	2	Research effects	In perpetuity
	Floodplains	Unmanaged fire	3	Research effects	In perpetuity
		Total ecological risk =	**0.21**		
MINESITE	Surface water	Uranium	4	Watching brief	2006
	Magela Ck	Sulfate	5	Watching brief	2006
		Magnesium	6	Watching brief	2006
		Manganese	7	Watching brief	2006
		Total ecological risk =	**0.00009**		
		Ra-226	8	Watching brief	2006
	Airborne/wind	Radon (Ra-222)	9	Watching brief	2011

NB: Ra^{226} and Ra^{222} (Radon) are included also

3.10.4 Future work

An analytical decision-making framework combining quantitative and qualitative ecological risk assessments for diffuse landscape and point source mining related threats is currently being developed. This approach is similar to a Bayesian

Network (BN) framework. Different degrees of belief associated with perceptions of risk, ranging from subjective expert opinion (for example, from park managers and traditional Aboriginal owners) to objective quantitative estimates derived from frequentist statistics (for example, the probability density functions reported here), can be integrated and the results communicated using simple influence diagrams and decision trees.

3.11 Tropical Rivers Inventory and Assessment Project (TRIAP)

3.11.1 Background

During 2005–06, the Department of the Environment and Heritage invested $0.3 million from the Natural Heritage Trust to fund the Tropical Rivers Inventory and Assessment Project (TRIAP), administered by Land and Water Australia's Tropical Rivers Programme, and managed by ***eriss***. The TRIAP commenced in late 2004, with the objective of establishing an information base for assessing change, undertaking ecological risk assessments of major pressures, supporting local and indigenous management, and strengthening holistic approaches for managing tropical rivers and their associated wetlands.

The project examines 51 catchments across northern Australia (from Broome in the west to the western tip of Cape York), covering some 1 192 000 km^2 (Figure 3.20). There are three focus catchments, representing each State or Territory within the study region, that are being assessed in more detail. These are the Fitzroy River in Western Australia, the Daly River in the Northern Territory, and the Flinders River in Queensland.

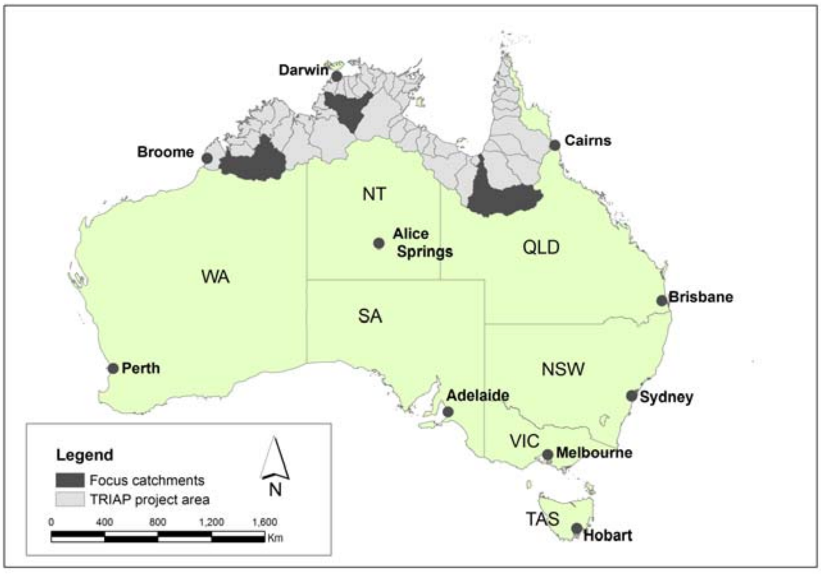


Figure 3.20 Location of Tropical Rivers Inventory and Assessment Project