Review of threats to biodiversity in the Northern Territory

Final report for NHT Project 2005/043

Extent	Severity 1	2	3	4	5
1	1	2	3	4	5
2	2	4	6	8	10
3	3	6	9	12	15
4	4	8	12	16	20
5	5	10	15	20	25

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December 2008

Natural Resource Management Board (NT) Inc.

Natural Heritage Trust
Helping Communities Helping Australia
An Australian Government Initiative

Northern Territory Government

Summary

This review of factors that threaten biodiversity in the Northern Territory ("NT") directly addresses Management Action 3-3 and Management Action Target 3-2 of the Integrated Natural Resource Management Plan (INRM Plan) for the NT:

> "*Undertake a major review of threatening processes, their environmental costs and the cost and feasibility of their control. Prioritisation of management options will be undertaken that takes into account the range of social, cultural and economic benefits that can be generated for conservation and resource management programs and agreements*".

> "*By 2010, a rigorous assessment of threatening processes, impacts, information gaps and costs of remedial actions is undertaken that informs the prioritisation of management options; and that informs the establishment of a systematic monitoring and management program for these threats*".

This project and report focuses on five factors that may most affect biodiversity in the NT: fire, feral animals, pastoralism, weeds and land clearing. Feral animals were subdivided into five kinds: large herbivores, rabbits, pigs, predators (cats and foxes), and cane toads, while 20 weed species were also considered separately, giving a total of 28 threats.

Five NT experts estimated the extent and severity of threats to five environmental values among five broad geographic regions (Savanna, Arnhem Land, Barkly, Arid and Southern) and 11 broad vegetation types, a process involving over 10,000 assessments. The process was objective, enabling threats and regions to be compared, but many of the individual values were subjective because the experts had to extrapolate from very limited information. Published information on the proven and likely threats to individual threatened species was also reviewed to provide an alternative regionally-based ranking.

The three factors assessed as having the most impact on the NT's biodiversity were large feral herbivores, buffel grass and pastoralism. Introduced pasture grasses comprised three of the top five threats and in the Savanna and Arnhem Land regions were the top three threats. At current levels, land clearing was ranked as a relatively low priority due to its limited geographic extent. Cane toads and feral predators were also ranked low because they generally do not affect vegetation condition or landscape function and only a restricted subset of native animal species. Fire was ranked as at an intermediate level, although in the analysis of threats to individual threatened species it was ranked first. The ranking of regions by total threats was Savanna (most impacted), Arnhem Land, Southern, Barkly and Arid. Rainforest was the most threatened vegetation type and mangroves the least.

The threat of climate change was not addressed directly, but it will have direct and substantial effects on biodiversity and will probably enhance the severity of the other threats considered here. Climate change places additional urgency on the need to act to manage threats to biodiversity.

The costs of attaining the Resource Condition Targets of the IRNM Plan were assessed for some threats. To do this, we estimated the resources required each year over a defined number of years. If the job will take many years, a discounting function was used to reduce the

present cost of such future work. These costs have been calculated illustratively for land clearing, fire and feral animals.

Feral herbivore control has an economic benefit in all regions except Arnhem Land (where it would cost $5m over 20 years). Assuming a modest carbon price (of $25 per tonne of CO_2 equivalent) is implemented in the future, it is more economical (over a 20-year timeframe) to retain native woodlands than to clear them for crops or pasture. Keeping cane toads from NT islands has a modest cost. Control of rabbits, pigs and cats is extremely expensive (under currently available control techniques, cats would cost more than $5 billion over 20 years). Fire management to target levels will cost more than $400 m over 20 years.

The cost of controlling threatening processes is very large and, even excluding cats and weeds, will require more than $1 billion over the next 20 years. However, some threats are much cheaper than others to control. Considering that feral herbivores are the highest-ranked threat and their control has a positive economic return, they are a clear example of a high priority management program.

In Appendix F, we provided some assessment of a range of threats to biodiversity not considered in detail in the main report and analyses. These threats comprised: climate change; disease; non-native invertebrates; “other” non-native vertebrates; exploitative use; changed hydrology; and tourism. In this section, we also map the incidence of different threats based on the number and distribution of listed threatened species.

In Appendix G, we identified a series of key information gaps that constrained the approach and execution of the modeling used here, the data it relied on, and/or the interpretation of its outputs. These were:

- The relative impacts of threats to at least some components of biodiversity are not well known;
- The impacts of some novel threats are not well known;
- There is little information for some current presumed threats (notably disease);
- The form of the threat/management response is not well known;
- We may be unaware of some biodiversity declines, that may be caused by threats not considered here;
- “Safe” levels of threat are poorly known (thresholds and limits);
- The responses of threats to climate change are poorly known;
- Social responses to threats are not well known and/or were not well incorporated into our economic models;
- The range of management options and techniques will change. Some current threats have no established control mechanisms;
- Investment in threat management should be geographically prioritised;
- Surveillance (for threats not yet present) was not considered in economic models;
- For many management actions, there is little information available on costs and efficacy.

For each identified gap, we provide recommended responses to address the shortcoming.

Contents

1. Introduction

Like all other regions of the world, the natural resources of the Northern Territory face a range of threats, many of which are costing government, business and individuals a great deal of money and effort to counter. The threats include fire regimes that are changing vegetation patterns, introduced animals that compete with or consume native wildlife, weeds that spread and replace native vegetation, and land uses that destroy or degrade vegetation. The scale of the problem can be gauged by the 15 mammal species that have become extinct in the NT since European settlement, and a total of 187 species of still-extant plants and animals that are now considered threatened (Woinarski *et al*. 2007).

The NT and Australian Governments continue to fund the mitigation of threats, and one of the main vehicles for funding is the Natural Heritage Trust. However, it is not clearly known whether the funding is being most efficiently and effectively targeted, either in terms of the threats or the locations where they are being tackled. In fact, a review of the first five years of the NHT was highly critical of the lack of strategy in the funding (Lowe 2005).

It would be very useful to have an objective assessment of which threats have the highest priority across the various regions and environments of the NT, and which can be addressed in the most cost-effective manner. Put simply, funding should be directed to threats and regions with the most severe impact and where there is the best chance of reducing the impact. This kind of analysis has never been attempted before in the NT. The Integrated Natural Resource Management Plan for the Northern Territory (INRM Plan, Anon 2005) was developed by the Landcare Council of the NT as a guide to what investment should be made on NRM issues in the NT. The Council acknowledged the need for a formal review of threats as one of the early actions under the plan, through Management Action Target 3-2:

> "*By 2010, a rigorous assessment of threatening processes, impacts, information gaps and costs of remedial actions is undertaken that informs the prioritisation of management options; and that informs the establishment of a systematic monitoring program for those threats*"

and Management Action 3-3:

> "*Undertake a major review of threatening processes, their environmental costs and the cost and feasibility of their control. Prioritisation of management options will be undertaken that takes into account the range of social, cultural and economic benefits that can be generated for conservation and resource management programs and agreements"*

This report provides much of that proposed review, for terrestrial environments.

In essence, our review provides a measure of the current severity of each threat for a range of environmental values it can affect, over a range of habitats and a range of geographic areas. This process requires the estimation of many values (over 10,000). The review is objective in that the values used at each stage of the assessment process are explicitly defined, repeatable, are applied in the same way for all threats, and use the best available scientific knowledge. The main advantage of this approach is that threats, vegetation types and regions can all be compared and prioritised as like-for-like. The main deficiency is in knowledge, because explicit and quantified evidence of the extent and severity of threats in regions and vegetation types rarely exists, and so the assessment relies on expert ‘extrapolation’ from the known to the likely.

The review focused on five of five factors known to affect biodiversity in the NT: fire, feral animals, pastoralism, weeds and land clearing. Partly because their impacts remain potential or not yet well defined, we do not consider other threats, such as disease, exotic invertebrates, or those arising from global climate change. Since feral animals and weeds comprise a range of species with differing effects, they were not be assessed as a single threat, but for these two factors the assessments were subdivided to individual pest species or groups of similar pest species. In the case of feral animals, they were split into groups with similar impacts - feral predators (cats and foxes); large feral herbivores (horse, donkey, buffalo, camel), rabbit, pig and cane toad. For the weeds, the 20 weeds with highest environmental risk (as ranked by the NT Weed Risk Management Committee, as at July 2007) were each assessed separately. This gave a total of 28 threats.

The NT was divided into five broad regions and eleven broad vegetation types and for each region / vegetation type combination an assessment of the impact of each threat on a range of values was made by a leading NT expert. Five impact values or attributes were assessed: vegetation condition, the abundance of threat-sensitive native species, landscape function, economic production and cultural values, although the last two were not used in this analysis. Each assessment was rated as a combination of the extent and the severity of the threat which were multiplied together to give an overall impact. Five extent classes were defined ranging from zero to more than 90% of the vegetation/region combination occupied. Similarly, five severity classes were defined, which differed for each attribute (see pp. 12-15), but all described the spectrum from healthy to seriously degraded.

When the assessments were complete, the cost of managing the threats was assessed. This was done in reference to the Resource Condition Targets (RCT) in the INRM plan. The RCT defines the target values for the extent and severity of each threat in the medium term future (2020). The cost is measured as how much money it would take to change the current state to the nominated target state. As with the original threat assessment, where possible, this was done for each vegetation/region combination and for each attribute and threat.

This review is largely based on the current context of the NT. Climate change is already with us, and the predictions for the future climate made by the IPCC are becoming ever more precise (IPCC 2001, IPCC 2007). The impacts on biodiversity will be substantial, although not yet tightly predictable. One of the problems is that the climate predictions are still uncertain, especially in terms of changes to rainfall patterns. For example, the latest modelling scenarios for the NT conclude that there could be more or there could be less total rainfall, although most likely there will be longer dry periods in either case (Hennessey *et al*. 2004). Another uncertainty is how climate change will affect the ways that species interact with each other. For example, a drying climate may have a negative influence on all species, but some will be relatively less affected than others, and may actually benefit from the decline of the others. This may be the case for some weeds and feral animals: they may actually increase under climate change. The other uncertainty is how society will respond to climate change. For example, it is unclear whether pastoralism will remain viable in a drying climate or even whether declining viability will result in lower or higher stocking densities. Climate change interacts with all of the other threats. Given the uncertainties and interactive nature of many of the impacts, climate change is not included here as a threat in its own right. Rather, we have attempted to take some consideration of likely climate change into our predictions of the future impacts upon biodiversity of the main individual threatening factors considered here.

The details of the review are described in the following sections. This begins with a brief description of each of the threats in Section 2. Section 3 details the methods and results of the threat assessment. Section 4 examines the published information on threats to listed threatened species, as a comparison to, and validation of, section 3. The costing process and results are described in Section 5. Section 6 synthesises the results and gives conclusions and recommendations. A detailed description of each of the threats is provided in Appendices 1-5 and the data generated by the assessment is in Appendix 6. For this report, we have provided economic costings for control of feral animals, fire and land clearing.

2. The Threats to Biodiversity

In this review, we have considered only five broad threatening processes: fire, feral animals, pastoralism, weeds and land clearing. These were considered by the review team to constitute the most significant of the threats to biodiversity in the NT. Note that many of these threatening processes may have complex interactions – for example, pastoralism may involve land clearing and then introduction of exotic grass species and consequential change in fire regimes. There are some obvious omissions from this list; principally mining, pollution, hunting (or harvesting) and disease. While mining is known to cause impacts where it occurs, the overall footprint is extremely small in comparison to the other threats in the list, and in any case the major impact on terrestrial biodiversity is via land clearing, which is already included. Pollution (including mine pollution) principally affects the aquatic environment and so is not considered here. Harvesting may affect a range of species, but this tends to be a localised issue. While disease may be a widespread and increasing threat, at present there is not enough information about wildlife disease in the NT to make any judgement about the extent of the threat.

Fire

Fire is a natural occurrence in all regions and environments in the NT, with the exception of some coastal and aquatic environments (e.g. mangroves). However, current evidence suggest that fire regimes have changed greatly since they were largely managed by Aboriginal people, with a shift to larger, more intense and possibly more frequent fire. Today, fires are less tightly or effectively managed and tend to occur as extensive, relatively intense wildfire under severe fire-weather conditions, either late in the dry season (Aug-Nov) in the savannas, or in spring-summer in central Australia. The impacts of these contemporary fire regimes vary greatly, differing with respect to types of habitat and the fire-response traits of individual species. For plants, for example, species possessing the capacity to resprout following burning (i.e. resprouters, like all eucalypts) are at a significant advantage in situations with frequent fire, compared with species which regenerate only from seed sources (i.e. obligate seeders, like many *Acacia* shrubs) when adult plants are killed.

Several habitat types are known to be particularly at risk from frequent burning - rainforests, heathlands, acacia shrublands and stands of the long-lived obligate seeder conifers, *Callitris glaucophylla* (desert cypress-pine) and *C. intratropica* (northern cypress-pine). Contemporary fire regimes are also implicated in the demise or decline of mammals (e.g. Bolton & Latz 1978, Woinarski *et al*. 2001) and granivorous birds (Franklin 1999). The spread of flammable introduced pasture grasses (e.g. gamba grass in northern savannas, buffel grass in central Australia) is likely to increasingly exacerbate problems associated with intense frequent fires.

Feral Animals

Exotic pest animals have major economic, environmental and social impacts across Australia (Commonwealth of Australia 2007). There are 19 species of exotic vertebrate pests in the Northern Territory (http://www.nt.gov.au/nreta/wildlife/animals/exotic/index.html). Donkey, horse, cane toad, Arabian camel, pig, water buffalo, fox and cat are considered major pests because they have a high level of overall impact at current densities and distributions. Other species such as the European rabbit, wild dog (excluding dingoes) and goat are considered to be moderate pests because they have lesser impacts, at current levels, to biodiversity. Other species like the house sparrow, rock pigeon, turtle dove, sambar deer, black rat, brown rat and banteng are considered minor pests as their overall impact is relatively minor.

Here we assess the level of threat posed by the donkey, horse, camel, water buffalo, pig, fox, cat, wild dog, rabbit and cane toad to biodiversity, production and cultural values in the Northern Territory. Because of differences in the mechanisms by which exotic pest animals affect the values under consideration, we grouped species with similar impacts in order to assess the level of threat. The threat groups were: feral predators (fox, cat, wild dog), large feral herbivore (donkey, horse, Arabian camel), rabbit, pig and cane toad. Note that all of the species considered here are recognised as serious threats to biodiversity, production and cultural values elsewhere in Australia, with the pig, fox, cat, rabbit and cane toad being listed under the Environment Protection and Biodiversity Conservation Act (1999) as key threats to biodiversity conservation in Australia.

Pastoralism

Pastoralism is the predominant land use in the Northern Territory in terms of areal extent, with c. 55% of the land area under some form of pastoral management (pastoral leases plus pastoral operations on some Aboriginal land tenures). Pastoral landuse spread through most suitable areas of the Northern Territory during the 1870s-1890s. The industry is now primarily based on breeding and turning off young store cattle for live export or fattening elsewhere in Australia; buffalo are farmed in small areas of the northern floodplains. Grazing is generally based on native pastures, although introduced species are used in some areas, and property and paddock sizes are generally very large (Oxley *et al.* 2005). Currently, increasing demand and rising costs, as well as high land values, are placing pressure on pastoralists to increase productivity, leading to further intensification of pastoral use (via infrastructure development, increased stocking rates and greater use of exotic pastures) (Ash *et al.* 2006).

Pastoral land-use affects ecosystem function and biodiversity through a number of mechanisms (James *et al.* 1999, Landsberg *et al*. 1999). Selective grazing by cattle alters plant species composition and vegetation structure, typically resulting in a reduction in the frequency of palatable perennial plants (or shrub invasion in some areas). Trampling leads to soil compaction and modifies infiltration rates and, in areas subject to heavy use (notably in riparian zones), to erosion. Habitat modification also has flow-on impacts on invertebrate and vertebrate fauna. Grazing impacts were initially highly concentrated around natural waters, however a major feature of more recent pastoral development in Australian rangelands has been the proliferation of artificial waterpoints, facilitating access by stock to a high proportion of most landscapes (Landsberg *et al.* 1997).

The spread of pastoral landuse throughout Australian rangelands has been implicated as a contributing factor to the decline and local, or total, extinction of some components of the biota - notably many arid-zone mammals but also some plant and bird species – although the precise mechanisms for this impact are undetermined (Woinarski & Fisher 2003). While some native species were rapidly extirpated in the initial ‘ecological shock’ of European colonisation, declines and local extinctions have continued since then (Burbidge & McKenzie 1989, Franklin 1999, Woinarski & Catterall 2005). Detailed studies in a variety of ecosystems (Landsberg *et al.* 1997, Fisher 2001, Fisher & Kutt 2006) have demonstrated that a portion of the biota (amongst most taxa studied) are ‘decreaser’ species that decline with increasing grazing pressure (or its correlate – decreasing distance to water points). A proliferation of waterpoints leads to the reduction of water-remote refugia for these species, and potentially their local or regional extinction (Biograze 2000). Conversely, ‘increaser’ species are favoured by grazing (or ready access to water), but these are frequently ‘weedy’ generalist species.

Grazing by stock contributes only a portion of the total grazing pressure in most NT landscapes, with additional grazing pressure from feral ungulates (buffalo, cattle, donkey, horse , camel) and native grazers (primarily macropods) (Fisher *et al.* 2004). Under this heading we concentrate on the threat posed specifically from grazing by stock, and related features of pastoral land use (such as development of artificial water points), as the impacts of feral animals are addressed separately, and the low density of macropods in most regions of the NT means that grazing pressure from this source is generally minor.

Land Clearing

Land clearing is widely accepted as one of the main threats to biodiversity, globally (Ehrlich 1988, Noss 1991) and in Australia (Williams *et al*. 2001). Relative to many other parts of Australia, there has been relatively little clearing in the NT - less than one percent of the vegetation has been cleared - but appreciably larger areas may be cleared in the future. The majority of the land clearing in the NT has occurred in the Daly Basin bioregion (approximately 10% cleared), and the hinterland of Darwin (approximately 6% cleared), and there are already measurable losses of biodiversity both from the cleared areas and from the remaining native vegetation in these regions (Rankmore & Price 2004).

Land clearing occurs for a variety of reasons, notably for agriculture, forestry, urban development and mining. Some of these actions have other impacts, particularly offsite impacts on water quality, but since aquatic biodiversity is not considered in this review, they can all be considered under the broad heading of land clearing.

Weeds

Naturalised foreign plants are recognised as major threats to biodiversity (and other values) across the Northern Territory (Smith 2002), and throughout the world. Their impacts can be diverse: at one extreme they may transform environments (for example, over very extensive areas, *Mimosa pigra* may change floodplain grasslands to impenetrable monospecific shrubland thickets), but they may also have impacts that are less extreme or less conspicuous, including alteration of fire regimes, reduction in seed and nectar resources for native animals, altering hydrology and soil properties, poisoning stock and native animals, and out-competing native plants (Fairfax & Fensham 2000).

In a recent review, Martin *et al*. (2006) listed 160 exotic plant species considered to be a current threat to Australia’s rangeland biodiversity. Each weed species is individual and each will have different impacts, extend over different areas, and be differentially capable of control. Hence it is not possible to readily compile a composite assessment of the biodiversity impact of weeds in general. Rather, here we assemble information and assessments for 20 different weed species. These were selected as those that rated most highly (for risk to biodiversity) in the Northern Territory’s weed risk assessment process (http://www.nt.gov.au/nreta/natres/weeds/risk: as at July 2007). It is recognised that other weed species may have at least local impacts on biodiversity in the Northern Territory, and that the Territory may be exposed in the future to additional significant weed species. The species included are listed in Table 1. Note that many of these species are recognised as serious threats to biodiversity elsewhere in Australia (and indeed elsewhere in the world). For example, Grice (2006) listed 15 principal weeds threatening biodiversity in the rangelands, and this list included 8 of the species considered here.

Table 1. Weed species considered in this review. Weed status: WONS= one of the 20 recognised weeds of national significance; 100 World's Worst=included in the list of 100 of the world's worst invasive alien species (Lowes *et al.* 2000); NT status: class A=to be eradicated; class B=growth and spread to be controlled; class C=introduction of species is prohibited.

Species	**Common name**	**Life form**	**Weed status**
Acacia nilotica	prickly acacia	shrub	WONS; Class A/C
Andropogon gayanus	gamba grass	grass	Class A/B
Azadirachta indica	neem tree	tree	
Cabomba caroliniana	Cabomba	aquatic herb	Class A/C
Cenchrus ciliaris	buffel grass	grass	
Cryptostegia grandiflora	rubber vine	vine	WONS; Class A/C
Hymenachne amplexicaulis	olive hymenachne	grass	WONS; Class B/C
Jatropha gossypiifolia	bellyache bush	shrub	Class B/C
Lantana camara	Lantana	shrub	WONS; Class B/C; 100 World's Worst.
Leucaena leucocephala	coffee bush	tree	100 World's Worst
Megathyrsus maximus	guinea grass	grass	
Mimosa pigra	Mimosa	shrub	WONS; 100 World's Worst; Class B/C
Parkinsonia aculeata	Parkinsonia	shrub	WONS; Class B/C
Pennisetum polystachion (and *P. pennisetum*)	mission grasses	grass	Class B/C
Prosopis spp.	Mesquite	shrub	WONS; Class A/C; 100 World's Worst
Salvinia molesta	Salvinia	aquatic herb	WONS; Class B/C
Schinus terebinthifolius	Brazilian pepper-tree	tree	100 World's Worst
Tamarix aphylla	athel pine	tree	WONS; Class B/C
Themeda quadrivalvis	grader grass	grass	Class B/C
Urochloa mutica	para grass	grass	

3. Estimation of threats

Methods

Review team

The review team consisted of experts on each of the threatening processes, plus a resource economist. Details of the members are shown in Table 2.

Table 2. Review team members

Person	Role	Relevant Publications
Owen Price, NRETA	Team Leader, Land Clearing	Rankmore BR and Price OF (2004). The effects of habitat fragmentation on the vertebrate fauna of tropical woodlands, Northern Territory. Pages 452-473 in D. Lunney, ed. *Australian Forest Ecology*. Royal Zoological Society of NSW, Mossman Price OF . Woinarski JCW, Milne D, Connors G, Harwood R and Butler M (2001) *A conservation plan for the Daly Basin Bioregion*. Report to NT Government, Parks and Wildlife Commission of the Northern Territory, Darwin. Price OF (2004). *A native vegetation retention strategy for the Darwin Region.* Department of Infrastructure, Planning and Environment, Darwin
Alaric Fisher, NRETA	Pastoralism	Woinarski JCZ and Fisher A (2003) Conservation and the maintenance of biodiversity in the rangelands. *Rangeland Journal* 25, 157-71. Fisher A, Hunt L, James C, Landsberg J, Phelps D, Smyth A and Watson I (2004) *Review of total grazing pressure management issues and priorities for biodiversity conservation in rangelands: A resource to aid NRM planning.* Desert Knowledge CRC and Tropical Savannas CRC, Alice Springs. Fisher A and Kutt A (2006). Biodiversity and land condition in tropical savanna rangelands:summary report. Tropical Savanna Management CRC, Darwin.
Jeremy Russell-Smith, NRETA	Bushfire	Russell-Smith J, Yates C, Edwards A, Allan GE, Cook GD, Cooke P, Craig R, Heath B and Smith R (2003) Contemporary fire regimes of northern Australia, 1997–2001: change since Aboriginal occupancy, challenges for sustainable management. *International Journal of Wildland Fire* 12, 283-97. Russell-Smith J and Edwards AC (2006) Seasonality and fire severity in savanna landscapes of monsoonal northern Australia. *International Journal of Wildland Fire* 15, 541–550
Glenn Edwards, Keith Saalfeld NRETA	Feral Animals	Edwards GP, Pople AR, Saalfeld K and Caley P (2004) Introduced mammals in Australian rangelands: Future threats and the role of monitoring programs in management strategies. *Austral Ecology* 29, 40-50. Edwards GP, Eldridge SR, Wurst D, Berman DM and Garbin V (2001) Movement patterns of female feral camels in central and northern Australia. *Wildlife Research* 28, 283 - 289
John Woinarski, NRETA	Weeds	Woinarski J, Mackey B, Nix H and Traill B (2007) *The Nature of the Northern Territory*. Australian National University E-press, Canberra. Woinarski JCZ and Fisher A (2003) Conservation and the maintenance of biodiversity in the rangelands. *Rangeland Journal* 25, 157-71.
Adam Drucker, CDU	Economics	Drucker AG and Latacz-Lohmann U (2003). Getting incentives right?: a comparative analysis of policy instruments for livestock waste pollution abatement in Yucatán, Mexico. *Environment and Development Economics*, 8, 261-284 Drucker AG, Gomez V and Anderson S (2001). The economic valuation of farm animal genetic resources: a survey of available methods. *Ecological Economics*, 36, 1-18

Regions

The five Grazing Land Management Zones defined by Fisher *et al.* (2004) (Fig. 1) were selected. These represent a combination of biophysical characteristics and land uses. These zones are an amalgamation of bioregions (Thackway & Cresswell 1995) and for this report we refer to them as Arnhem Land, Savanna, Barkly, Southern and Arid.

Vegetation Types

As with regions, a small set of vegetation types was required to keep the combination set manageable. We used an 11 class amalgamation of the 1:1 000 000 vegetation map of the NT (Wilson *et al.* 1990) to define broad vegetation types (Table 3, Fig. 2). Note that these vegetation types vary greatly in extent, from Spinifex grasslands that occupy 38% of the NT to rainforests that occupy less than 0.1% (Table 4). Likewise, the vegetation types are not all present in each region (Table 4). There are some limitations with the 1:1000 000 vegetation map, principally that it does not recognise any heath vegetation in Arnhem Land. To counter this problem we used more precise mapping of heathlands in Arnhem Land from Blake (2005), with a total area of 24,510 km^2.

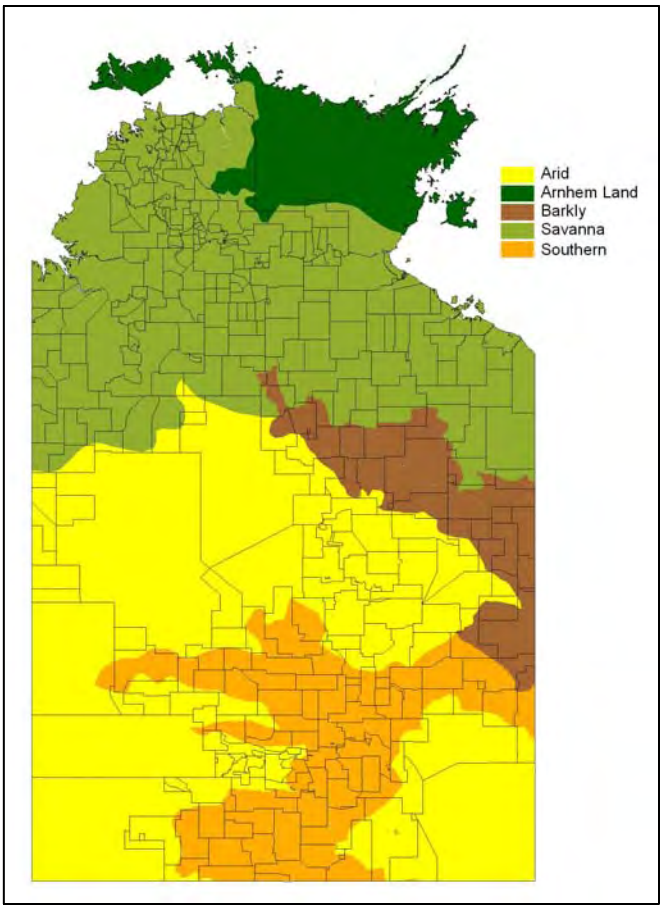


Figure 1. Regions (Grazing Land Management Zones – Fisher *et al.* 2004)

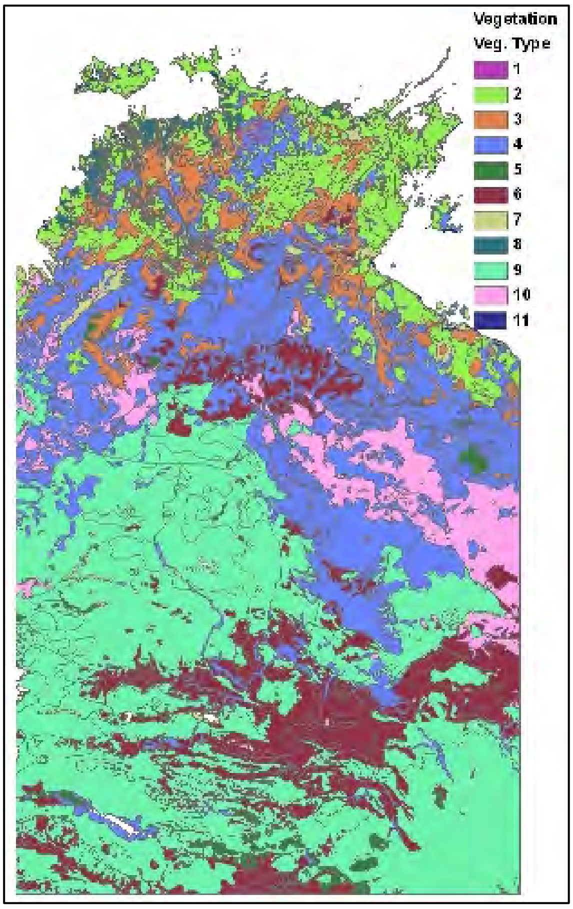


Figure 2. Vegetation types (refer to Table 3 for a description of types).

Table 3. Vegetation categorisation used in this review (derived mostly from the 1:1 000 000 vegetation map of Wilson *et al*., 1990). "JRS group" refers to rainforest types identified by Russell-Smith (1991).

No	Vegetation category	Habitat types from wildlife surveys description	Wilson *et al.* mapunits
1	Rainforest and riparian	Coastal vine forest (JRS group 9)	1 (part)
		Riparian rainforest (JRS groups 10,11)	1 (part)
		Spring rainforest (JRS groups 1-6,13)	1 (part)
		Allosyncarpia forest (JRS groups 7,8)	2
		dry (non-coastal) thicket (JRS groups 12,14-16)	1 (part)
		Euc. camaldulensis riparian strips, and/or with *Terminalia platyphylla, T. bursarina, Syzygium eucalyptoides eucalyptoides* and *Lophostemon*.	
2	Eucalypt forests	*Eucalytus miniata – E. tetrodonta* forest (tussock grass understorey) {sometimes with *Corymbia. bleeseri* &/or *Callitris*}	3-9,11,14
		Eucalytus miniata – E. tetrodonta forest (hummock grass understorey) {sometimes with *Corymbia. bleeseri* &/or *Callitris*}	10,12,13
3	Seasonally inundated woodland	*Corymbia* woodland (dominated by any of *C. tectifica, C. grandifolia, C. latifolia, C. bella/arafurica/papuana*), typically on seasonally wet flats {sometimes dominated by *Erythrophleum* and/or *Lophostemon*}	15,16,18 (part)
		Eucalyptus ptychocarpa wet sandy areas, typically with *Pandanus, Lophostemon*	
		Barringtonia dominated margins of billabongs,	
4	Eucalypt woodlands	Slope & hill euc. woodland dominated by *C. dichromophloia* (s.l.), *C. tintinnans, Euc. kombolgiensis, E. aspera* and/or *E. phoenicea* (tussock grass understorey).	17,20 (part)
		Slope & hill euc. woodland dominated by *C. dichromophloia* (s.l.), *C. tintinnans, Euc. kombolgiensis, E. aspera* and/or *E. phoenicea* (hummock grass understorey)	20 (part),31-34
		Sandy woodland dominated by *C. polycarpa* (tussock grass understorey)	18 (part)
		Sandy woodland dominated by *C. polycarpa* (hummock grass understorey)	
		C. terminalis or *C. opaca* woodland (tussock grass understorey)	19,22
		C. terminalis or *C. opaca* woodland (hummock grass understorey)	41,42
		E. pruinosa woodland (tussock grass understorey)	23, 45 (part)
		E. pruinosa woodland (hummock grass understorey)	39 (part)
		E. microtheca woodland, including *Excoecaria parvifolia* dominated open woodland	24-28
		Woodlands dominated by *E. leucophloia* or *E.brevifolia* (usually have hummock grass understorey)	35-38
5	Heath / Chenopod	*Lysiphyllum cunninghamii* open woodlands {often with *Hakea arborescens, Grevillea striata}*	39 (part), 45 (part), 46
		Sandy heathlands and open woodlands (including some of *Corymbia ferruginea* (s.l.), *Jacksonia* spp., *Banksia*, *Asteromyrtus* spp., *Acacia* spp, *Grevillea* spp.) (tussock grass understorey)	48,51 (part), 102 (part)
		Sandy heathlands and open woodlands (including some of *Corymbia ferruginea* (s.l.), *Jacksonia* spp., *Banksia*, *Asteromyrtus* spp., *Acacia* spp, *Grevillea* spp.) (hummock grass understorey)	40
		Chenopod shrublands	108-111
6	*Acacia*	*Acacia shirleyi* woodland	55,56
		Macropteranthes kekwickii low woodland	57

		Acacia aneura woodland/shrubland	58,60, 65, 66, 69-71
		Acacia georginae woodland	62-64
		Terminalia arostrata open woodland	44
6	*Acacia*	other *Acacia* woodlands and thickets on sand (tussock grass understorey) other *Acacia* woodlands and thickets on sand (hummock grass understorey)	47 67,68,72,73,74
7	*Melaleuca*	Low woodland/shrubland dominated by *Melaleuca minutifolia, M.citrolens* or *M. acaciodes* Low woodland/shrubland dominated by *Melaleuca nervosa* or *M. viridiflora* (+/- *Petalostigma banksii*} *Melaleuca* forest/woodland	49,50,106 (part) 51 (part) 53
8	Floodplain	sedgeland on floodplain grasslands on floodplain	54 (part) 54 (part)
9	Spinifex	*Triodia* hummock grassland	75-94
10	Grassland	*Astrebla* grassland Tall grasslands (typically with *Dichanthium, Chrysopogon, Sorghum*) Other short grasslands (typically with *Aristida, Eragrostis, Enneapogon* and/or *Xerochloa*) Bluebush swamps	96,97 98 99,100,101 (part), 104 107
11	Mangroves and coastal	Mangroves strand (*Casuarian equisetifolia*, etc) and coastal *Vetiveria* grasslands	105 102 (part), 103

Table 4. The extent of each vegetation type in each region (km^2).

Vegetation type	Arid	Arnhem Land	Barkly	Savanna	Southern	Total
Rainforest and riparian	0	771	0	258	0	1029
Eucalypt forest	0	41269	0	69847	0	135626
Inundated woodland	0	12388	0	69354	0	81742
Euc. woodland	71953	14479	17283	189332	9422	302469
Heath/chenopod	5671	24510	204	4522	6945	17342
Acacia	39632	444	11692	28634	96249	176651
Melaleuca	1529	4373	0	16758	0	22660
Floodplain	0	1320	0	7418	0	8738
Spinifex	429165	0	1984	13377	67767	512293
Grassland	2182	0	63241	23230	842	89495
Mangrove/Coastal	0	1422	0	949	0	2371
Total	550132	100976	94404	423679	181225	1350416

Climate Change Scenario

Hennessey *et al.* (2004) modelled the likely changes to climate in the NT and produced scenarios for the years 2030 and 2070. While there is considerable uncertainty in these predictions, it is likely to be warmer and with more severe dry periods, especially in the arid zone. In this review we used the year 2030 predictions, rather than 2070, to keep the priority setting more in line with the resource condition targets in the INRM. The team decided to address climate change in the discussion rather than repeat the entire matrix-filling exercise and to place emphasis on predictions about rainfall seasonality in the scenario.

Assets

Five values or assets were selected to capture the range of impacts that each threat may have: Vegetation Condition, Threatened & Sensitive Species, Landscape Function, Production and Cultural Values (Table 5). The first three of these are biodiversity assets, in that they relate directly to the health of species or the environment supporting species. The other two are human values and were included to reduce possible conflict in decision making. For example, where an introduced pasture grass has a positive effect on pastoral production, this is recorded so that tradeoffs can potentially be explored in the costing section of the review. In the analysis comparing the level of threat, only the three biodiversity assets were used - the other two assets are referenced in the discussion.

The <u>extent and condition of native vegetation</u> is a fundamental attribute describing healthy ecosystems: all species depend on vegetation for habitat and food sources. Unfortunately, vegetation condition is difficult to define in absolute terms because different species perceive vegetation in different ways: what is good for one species may not be good for another. Nevertheless, some conceptual classifications capture well the range of conditions that are likely to reflect the proportion of the original biota remaining. In this review, we used the VAST scheme (Thackway & Lesslie 2006), now widely recognised and used to describe, classify and map vegetation condition across Australia..

<u>Threatened and susceptible species</u> refers to direct ecological impacts that may not be captured by vegetation condition. For example, cats may eliminate some species from some areas without any change in vegetation condition. Here the term ‘susceptible’ refers to any species that are known to be affected by that particular threat (not just listed threatened species). The impact is assessed as the proportion of these species that are likely to decline in, or be lost from, a landscape experiencing the threat.

<u>Landscape function</u> (Ludwig *et al.* 1997) is a physical measure that is useful for capturing environmental impacts that may not be evident using the two indicators above. It is a measure of how well processes such as water and nutrient capture are functioning. If landscape function declines (for example in an over-grazed landscape), then ecological impacts can be predicted even if the specifics of the exact nature of the impact of pastoralism on biota is not known. There are well established field methods for measuring landscape function (http://www.cse.csiro.au/research/efa/).

<u>Production</u> reflects the economic activity derived from each of the vegetation/region combinations, which in most cases is from cattle pastoralism but also includes tourism and harvesting. Activities such as mining and manufacturing are not here factored into this value because they are largely independent of biodiversity.

Cultural values refer to those human values that are not expressed in economic terms. These include Aboriginal spiritual values and Aboriginal access to land and food sources, and aesthetic and recreational values for the broader community. We used established texts, where they are available, to estimate these values, but note that we consider our assessments for this character were limited.

Extent scores for assets

For each region/vegetation combination, a score was entered for the extent of the threat on a scale of zero to five, being 0: <5%, 1: 5-10%, 2: 10-25%, 3: 25-50%, 4: 50-90% and 5: >90% of the area occupied. This value was the extent currently occupied by the threat, not the future potential. There are some weed species that are currently uncommon in the NT but have the potential to become serious problems in the future (e.g. Rubbervine). These are not well catered for in this assessment.

Severity scores for assets

For each of the five assets, we need to define a set of classes describing their condition: the state that a threat may place them into. For vegetation condition, it was decided that the VAST definitions gave too little discrimination when there was a minor (but significant) impact, so that class 2 was subdivided into 2a and 2b (Table 6). For landscape function, definitions were derived from work for the National Land and Water Resouces Audit (Table 7).

Table 5. Assets and severity classes (see Tables 6 and 7 for more details on vegetation condition and landscape function).

Asset	Severity Classes
Biodiversity	
Vegetation Condition	VAST classes 1-5 (modified from 0-4 with class 2 subdivided) 0 is pristine, 5 is replaced by exotics
Threatened and susceptible species	0 = no impact, 5 = major loss of a whole suite of susceptible and threatened species, 1 – 4 represent stages on this scale.
Landscape Function	0: Insignificant change 1: Minor 2: Moderate 3: Major 4: Catastrophic
Socio-Economic	
Production	0: <2% impact on Gross production 1: 2-10% 2: 10-30% 3: 30-70% 4: >70% impact
Cultural Values	0: Negligible impact 1: Obvious but minor 2: up to 1/3 of value gone 3: 1/3 - 2/3 of value gone 4: Essentially no value left

Table 6. Severity classes for vegetation condition, derived from the VAST scheme (Thackway & Lesslie 2006). 'State' is the original VAST class and 'Class' is the severity class used in this analysis.

State	Class	Description	Management
1: Residual	0	Native vegetation community, structure, composition and regenerative capacity intact- no significant perturbations from land use/ land management practice.	Uncleared, Ungrazed, Natural fire regimes, No weeds
2: Modified		Native vegetation community, structure, composition and regenerative capacity intact- perturbed by land use/ land management practice.	Sustainable grazing, selective logging, non-natural fire regimes
2a	1	Minor alteration in species composition, size structure and/or dominance	Light grazing; fires too frequent, no weeds
2b	2	Moderate alteration in species composition, size structure and/or dominance	Moderate grazing, fires too frequent and intense, some weeds
3: Transformed	3	Native vegetation community, structure, composition and regenerative capacity significantly altered by land use/ land management practice.	Heavily grazed, trees thinned to promote pasture, weedy
4: Replaced – Adventive	4	Native vegetation replacement – species alien to the locality and spontaneous in occurrence.	Severe weed invasion, natives in the minority
5: Replaced - Managed	5	Native vegetation replacement with cultivated vegetation	Improved pasture, crops, plantations

Table 7. Severity classes for Landscape Function (adapted from Ludwig *et al*. 1997).

Class	Description
0	**Insignificant**. May be some seasonal reduction in ground cover, but no reduction basal area of perennial plants or fine-scale patchiness. Changes to soil surface minor and localised. No increase in runoff.
1	**Minor**. Seasonal reduction in ground cover, but only small reduction in basal area of perennial plants. Minor compaction of soil surface but cryptagamic crust mostly retained. Minor localised erosion on drainage lines.
2	**Moderate**. Substantial seasonal reduction in ground cover and significant reduction in basal area of perennial plants. Significant widespread compaction of soil surface, reduction in macropore density and disruption of cryptagamic crust. Moderate erosion along drainage lines.
3	**Major**. Substantial long-term reduction in ground cover and major reduction or removal of perennial plants. Substantial compaction of soil surface and removal of cryptagamic crust and/or patchy removal of topsoil layer. Widespread erosion along drainage lines, cutting back into runoff areas.
4	**Catastrophic**. Widespread removal of ground cover including perennial plant layer. Widespread disruption to soil profile and removal of topsoil. Major erosion of drainage lines with widespread gullying and/or scalds.

Impact Score (Impact = Severity x Extent)

The approach we have adopted is a modification of the Risk Assessment model, where a matrix of likelihood against consequence is used. For this project, we use Severity against Extent, where Severity is the severity class that a threat puts an asset into, and Extent is the percentage of the region that is affected. Each cell in this matrix was assigned an Impact score (Low, Moderate, High, Very-high, see Table 8) by multiplying extent and severity. Since landscape function had only four non-zero severity classes, the values were transformed by multiplying by 1.25 to make them congruent with vegetation condition and threatened & sensitive species. Very-high represents a score of at least 9, which can come about from several combinations - such as a threat that has a minor impact at any location where present but that it is present over most of a region, or one that essentially replaces the native species but has a minor extent, or one that is moderate in both extent and severity.

The main focus of this review is in the impact on biodiversity, so the three biodiversity assets (vegetation condition, species, and landscape function) were used in the numerical analysis. However, as an adjunct we also report on the cultural impact of the 28 threats. The impacts on production are an integral part of the costing section, and so are not reviewed in this section.

Table 8. Impact score (Severity * Extent). The table is colour coded into areas of increasing impact (grey = low, pale orange = moderate, orange = high, red = very-high).

	Severity				
Extent	1	2	3	4	5
1	1	2	3	4	5
2	2	4	6	8	10
3	3	6	9	12	15
4	4	8	12	16	20
5	5	10	15	20	25

Comparison of Threats

The impact scores can be presented in a variety of different ways. The most useful were to analyse the scores to reveal the most severe threats and the most threatened regions and vegetation types. The threats were compared by calculating the mean impact score across assets and regions for each vegetation type, and the mean impact score across assets and vegetation types for each region. These calculations were area-weighted, so that, for any particular threat, the contribution of each vegetation type to the score for a region is the product of the threat score for that vegetation type and its percentage area in the region. The result is two tables that can be used to rank the threats, the regions and the vegetation types. For these calculations, only the three biodiversity assets were used.

Results

Populating the tables

A report for each threat is attached in the appendices, and each gives some information on the assumptions made when populating the tables and the problems encountered.

Comparison of threats

When the sum of all threats is considered on a regional comparison, the Savanna Region has the highest total level of threat, followed by Arnhem Land, Southern, Barkly and Arid (Table 9).

The highest ranked threatening processes across all regions were large feral herbivores, pastoralism and buffel grass . The highly ranked threats were those that had (at least) high impact across several regions. For the Savanna region, the top three threats were mission grass, gamba grass and large feral herbivores; for Arnhem Land they were mission and gamba grasses and fire; for the Barkly region they were prickly acacia, parkinsonia and pastoralism; for the Southern they were buffel grass, large feral herbivores and pastoralism; and for the Arid region they were large feral herbivores, buffel grass and feral predators.

Feral predators and cane toads were ranked relatively low (10 and 15), because these threats generally do not affect vegetation condition or landscape function (and hence cannot score highly on our combined impacts) and only a subset of native animal species.

There were several threat/region combinations that had an impact rated as very high (i.e. threat scores of 9 or higher). These were the impacts of prickly acacia and parkinsonia in the Barkly region, of mission grass in the Savanna region and buffel grass in the Southern region. Notice that these values are a mean across all vegetation types and the three biodiversity assts (vegetation condition, sensitive species and landscape function), so to achieve values as high as 9 implies a major impact on many environmental aspects in many vegetation types.

There was a large range of values for the overall impact of the threats, with 10 of the threats scoring 0.4 or less - which is 10 times less than the top ranked three threats. These low scoring threats included land clearing, rubbervine and cabomba. Land clearing had a low score because, even though the severity score was very high, the extent was localised in comparison to the large regions used here. In other words, from an NT perspective, land clearing is not a priority threatening process, even though in certain sub-regions (Darwin and Daly) it may be a high priority. Rubbervine and bellyache bush are not currently extensive in the NT and this accounts for their low score, but they threaten to become much more extensive if they are not controlled. Cabomba and salvinia are aquatic weeds which occupy a very small extent in the NT. Given that this review considers the entirety of the NT at a coarse scale, and that it considers mainly terrestrial issues, it is not surprising that they get low impact values, and they would more appropriately be dealt with in a specific review of threats to aquatic ecosystems.

When the results were displayed for vegetation types (averaged across regions), the three most threatened vegetation types were rainforest/riparian, floodplains and Melalueca, while heaths and mangroves had lowest total threat. This ranking reflects the tendency for weeds and feral animals to concentrate in the wetter parts of the environment. The top three ranked individual threats were large feral herbivores, pastoralism and mimosa (Table 10). Mimosa was in this list and not the previous one because it has a major impact on several wetland vegetation types, which are themselves small in extent and so do not contribute greatly to the regional perspective. For similar reasons, pigs ranked highly in this analysis (number five) because of their high impact on restricted mesic vegetation types. Conversely, buffel grass has a lower rank in this analysis because it has a broad extent but (at present) is having a major impact in relatively few vegetation types. In this table, fire is ranked number 4 while land clearing and rubbervine are still very low on the list, but cabomba and salvinia are ranked higher than in Table 9 because they have major impacts on one or more vegetation types. In general, this scheme tends to increase the priority of threats that occur mostly on the more restricted vegetation types, as all vegetation types get equal weighting irrespective of their areas.

To conclude, in the absence of a measure of cost effectiveness (which will be covered in Section 4), the priorities for action in the NT as a whole are better management of pastoralism, control of large feral herbivores and control of mimosa and introduced pasture grasses.

For particular regions the priorities vary slightly, with prickly acacia and parkinsonia being priorities in the Barkly region, and feral predators a priority in the Arid region.

Perhaps unexpected was the finding that pastoralism, when combined with the spread of exotic pasture grasses, is responsible for the greatest pressure on our native wildlife, while more widely recognised threats such as cane toads and fire received relatively low scores.

Table 9. Threat values for each region, ranked according to the mean across all regions. The top 5 threats in each region are highlighted.

Threat	Savanna	Arnhem Land	Southern	Barkly	Arid	Mean
Large Herbivore	**5.1**	**2.44**	**8.31**	0	**8.33**	4.84
Pastoralism	4.3	0	**7.84**	**8.97**	**2.63**	4.75
Buffel Grass	2.22	0.56	**10.33**	**4**	**6.6**	4.74
Mission Grass	**10.61**	**7.09**	0	0.18	0	3.58
Fire	**4.87**	**5.99**	3.81	0.21	0.78	3.13
Prickly Acacia	1.55	1.23	0.61	**11.95**	0.15	3.1
Gamba Grass	**7.67**	**6.67**	0	0	0	2.87
Parkinsonia	1.52	0.49	0.02	**10.36**	0	2.48
Predator	0.58	0.54	**4.88**	0	**4.76**	2.15
Small Herbivore	0	0	**6.11**	0	**3.62**	1.95
Mesquite	1.32	0.86	0.55	**6.46**	0.08	1.85
Guinea Grass	**4.49**	**3.65**	0	0	0	1.63
Pig	2.42	2.37	0	0	0	0.96
Grader Grass	2.22	1.85	0.05	0.18	0	0.86
Cane Toad	1.57	2.04	0	0	0	0.72
Mimosa	1.94	1.63	0	0	0	0.71
Para Grass	1.32	1.09	0	0	0	0.48
Olive Hymenachne	1.14	0.93	0	0	0	0.41
Lantana	1.03	0.98	0	0	0	0.4
Clearing	1.36	0.02	0	0	0	0.28
Coffee Bush	0.66	0.66	0	0	0	0.26
Neem	0.62	0.57	0	0	0	0.24
Salvinia	0.51	0.53	0	0	0	0.21
Brazilian Pepper	0.34	0.61	0	0	0	0.19
Cabomba	0.21	0.25	0	0	0	0.09
Bellyache Bush	0.37	0.02	0	0	0	0.08
Rubbervine	0	0.06	0	0	0	0.01
Athel Pine	0	0	0	0	0	0
Total	**59.94**	**43.13**	**42.51**	**42.31**	**26.95**	

Table 10. Threat values for each vegetation type, ranked according to mean across all vegetation types. Vegetation types are arranged according to total threat score.

Threat	Rain-forest	Flood-plain	Melal-euca	Euc forest	Euc. Woodl.	Inund. woodl	Grass-land	Acacia	Spinifex	Heath/ Chen.	Mang-rove	Sum
Large Herbivore	6.16	6.53	6.47	2.67	4.54	3.27	2.88	6.82	9.23	2.3	0.2	4.64
Pastoralism	2.67	5.94	4.01	1.55	3.87	4.24	9.43	8.11	2.46	3.23	0.53	4.19
Mimosa	2.33	17.33	10.26	0	0	7.33	0	0	0	0	1	3.48
Fire	4.67	6	4.66	5.74	4.11	0	0	3.17	1.56	6.27	0.67	3.35
Pig	8.33	4.67	9.52	1.91	1.35	4	0	0	0	0	2.67	2.95
Mission Grass	4.67	0	0.93	11.89	10.65	1	1.21	0.77	0.03	0.69	0	2.89
Buffel Grass	0	0	0.14	2.02	5.55	0	3.72	11.88	4.86	0.3	0	2.59
Para Grass	0	16	6.53	0	0	4.67	0	0	0	0	0	2.47
Parkinsonia	0	3.96	3.45	0	0.27	4.57	11.83	0.31	0	0	0	2.22
Gamba Grass	4.67	0	0	9.83	8.09	0	1.21	0.16	0	0	0	2.18
Salvinia	9	6	5.6	0	0	1	0	0	0	0	0	1.96
Olive Hymenachne	1	11	4.35	0	0	4.67	0	0	0	0	0	1.91
Mesquite	2	2	1.93	0	0.06	6	7.46	0.9	0	0	0	1.85
Cane Toad	4	5	2.49	1.84	0.96	2.67	0	0	0	0.04	2	1.73
Prickly Acacia	0	0.85	0.74	1.64	1.85	1	12	0.81	0	0	0	1.72
Predator	1	0.33	0.31	0.55	1.35	0.33	0.2	3.96	4.88	1.74	1.33	1.45
Cabomba	11	0	0.93	0	0	1	0	0	0	0	0	1.18
Guinea Grass	1.67	0	0	5.7	4.94	0	0	0	0	0	0	1.12
Brazilian Pepper	9	0	0	0.82	0	1	0	0	0	0	1	1.07
Lantana	6.75	0	0	1.37	1.12	0	0	0	0	0	1	0.93
Small Herbivore	0	0	0	0	0.02	0	0	5.13	4.23	0.6	0	0.91
Coffee Bush	6	0	0.93	0.82	0.67	0	0	0	0	0	1	0.86
Rubbervine	8	0	0	0	0	0	0	0	0	0	0	0.73
Grader Grass	1	0	0	2.73	2.33	0	0.87	0	0	0	0	0.63
Clearing	0	0	0	2.06	0	3.39	0	0.01	0	0.43	0	0.54
Bellyache Bush	3.25	0	0.74	0	0	1.7	0	0	0	0	0	0.52
Neem	3.09	0	0	0.82	0.67	0	0	0	0	0	0	0.42
Athel Pine	0	0	0	0	0	0	0	0	0	0	0	0
Total	**100.26**	**85.61**	**63.99**	**53.96**	**52.4**	**51.84**	**50.81**	**42.03**	**27.25**	**15.6**	**11.4**	

Cultural Impact

The rankings for Cultural impact contrasted with those for biodiversity. Feral predators and cane toads ranked most highly (although note that no cultural impact was assigned for pastoralism). Arnhem Land was the most threatened region and the Arid region the least.

Table 11. Threats to cultural values.

Threat	Arnhem Land	Savanna	Barkly	Southern	Arid	Mean
Feral Predator	2.77	2.48	4.94	8.61	8.85	5.53
Cane Toad	11.68	6.67	0	0	0	3.67
Prickly Acacia	1.72	1.55	8.65	0.61	0.14	2.53
Large Herbivore	2.22	0.55	0	3.31	5.11	2.24
Mission Grass	4.98	5.5	0.18	0	0	2.13
Gamba Grass	4.8	3.85	0	0	0	1.73
Fire	2.42	2.32	0.24	1.91	0.78	1.53
Parkinsonia	0.25	0.77	5.97	0.02	0	1.4
Mesquite	0.43	0.66	4.45	0.55	0.08	1.23
Rabbit	0	0	0	2.72	1.78	0.9
Guinea Grass	2.4	1.84	0	0	0	0.85
Buffel Grass	0	0	0.37	2.35	0.83	0.71
Grader Grass	1.6	1.33	0.18	0.05	0	0.63
Mimosa	1.37	1.62	0	0	0	0.6
Pig	1.26	0.79	0	0	0	0.41
Olive Hymenachne	0.79	0.97	0	0	0	0.35
Coffee Bush	0.88	0.66	0	0	0	0.31
Lantana	0.84	0.62	0	0	0	0.29
Neem	0.8	0.61	0	0	0	0.28
Para Grass	0.53	0.66	0	0	0	0.24
Brazilian Pepper	0.84	0.33	0	0	0	0.23
Salvinia	0.53	0.51	0	0	0	0.21
Clearing	0.01	0.68	0	0	0	0.14
Cabomba	0.23	0.21	0	0	0	0.09
Bellyache Bush	0.02	0.37	0	0	0	0.08
Rubbervine	0.02	0	0	0	0	0
Athel Pine	0	0	0	0	0	0
Pastoralism	0	0	0	0	0	0
Total	**43.39**	**35.55**	**24.98**	**20.13**	**17.57**	**28.31**

Climate Change

It is very difficult to predict how the threats to biodiversity will change under climate change scenarios, or how the biota will respond to those threats. This is partly because the future climate predictions are uncertain and partly because the response of the biota and human society will be complex and unpredictable. For each of the threats, reasonable arguments can be made that their severity will decrease, and conversely that they will increase.

Assuming that carbon will have a price in the near future, it may well become uneconomical to clear land for pastoralism or cropping, as the analysis in Appendix 1 shows. Carbon pricing may also have a negative impact on the pastoral industry because of the large amounts of methane produced by cattle. A decline in pastoral activity in the NT may have positive effects on biodiversity (because one of the threatening processes is reduced), but may have negative effects too (because there would be fewer land managers controlling some of the other threatening processes). If, as is likely, the NT becomes more arid this will also put pressure on the viability of pastoralism in at least some areas. However, this may not lead to less impact, especially if pastoralists do not respond quickly to the changes. If mean annual rainfall declines but cattle numbers are not reduced then the ecological impacts will increase.

Changes in the rainfall pattern may shift the areas of the NT that are suited to different agriculture in unpredictable ways. There are also global factors, such as the possible collapse in production in some of the major food-growing regions of the world, which will increase food prices and make farming on the relatively marginal NT soils more economical. All of these factors are hard to predict and are likely to interact.

Feral herbivores are likely to become less abundant due to reduced rainfall, but, as with managed cattle, there could be even greater impact as they compete with native species for reduced primary production. Feral cats and foxes are known to decrease in abundance during drought, as their populations track those of the native fauna (with a time lag). Again the impact of climate change is hard to predict since they may have a greater impact in the times when a post-rain boom is coming to an end.

Irrespective of the changes in the threats, climate change brings major direct threats to biodiversity, expressed chiefly in overall warming and longer dry periods in the NT. The biota will be forced to adapt or to move geographically to track their favoured climatic envelope. The threatening processes serve to compound the problems facing the biota. For example, land clearing, when it fragments landscapes, makes far more difficult for many species to migrate to new areas.

4. Estimation of threats from threatened species information

Introduction

There are 203 native species listed as threatened in the NT under the *Territory Parks and Wildlife Conservation Act* and/or *Environmental Protection and Biodiversity Conservation Act*. As part of an ongoing program to attempt to conserve these species, the NT Department of Natural Resources, Environment and the Arts has reviewed the processes threatening each species and published the results in a book "Lost From Our Landscape: Threatened Species in the Northern Territory" (Woinarski *et al*. 2007). The assessments of the threatening processes vary in their precision from the results of detailed scientific studies, to expert opinion based on very scant information. Nevertheless, the body of information provides a very useful insight into the processes driving the decline of native species. It should be highlighted that this analysis is not directly comparable to the process-based review of Section 3 because the data were not generated for this purpose and it focuses on a narrow suite of species rather than biodiversity in its entirety. Section 3 is the source of the major conclusions in this report, while this analysis is used to provide additional insights and as a form of cross-validation.

Method

The review includes information on the known distribution of each species, based on wildlife atlas records. We used this information to identify in which regions each species occurs, and combined this information with the listed threats for each species to calculate the most common threatening process in each region.

The threats used in this analysis are slightly different from those used in the main review. Some threatened species are threatened by specific processes that do not figure significantly across all species or regions. These additional threats include mining, harvesting and disease. In addition, some threats are not a result of human intervention. The most common of these was "stochastic processes" - referring to the risk that species with naturally small populations will become extinct as a matter of chance events.

Results

Fire was the main threatening process identified in most regions and overall (Table 12). This result contrasts to the review of section 3, where fire was ranked lower. The high priority given to fire in this analysis suggests that it be given more consideration than Table 9 indicates.

Weeds are the second highest ranked threatening process and this accords well with the main review. Land clearing is ranked low and this also accords with the main review. It is also notable that feral predators score highly in this tabulation. This discrepancy with Table 9 probably arises because feral predators have minimal impacts on vegetation condition or landscape function, and so failed to score highly in the main review. As with fire, Table 12 suggests that feral predators should be given more consideration than indicated in the other tables.

Table 12. Threat rankings calculated from published threats to threatened species (Woinarski *et al.* 2007). The columns are regions and the rows are threats, both in descending orders of the degree of threat. Note that the sum of all values exceeds the number of threatened species considered, because more than one threat may affect any individual species.

Threat	Savanna	Arnhem	Arid	South.	Marine	Barkly	Total
Fire	43	40	48	34	0	9	136
Weeds	18	19	33	18	0	3	70
Feral Herbivores	15	23	13	10	0	3	61
Feral Predators	15	10	15	19	7	6	58
Pastoralism	17	9	22	18	0	4	58
Stochastic	17	26	5	3	0	0	43
Clearing	19	16	5	5	4	2	36
Hydrological Change	14	16	4	4	0	2	30
Harvesting	4	5	6	7	6	3	20
Pesticides	3	2	2	1	10	1	17
Fishing	2	0	0	0	12	0	14
Toads	10	4	0	0	0	2	11
Climate Change	5	2	2	2	0	0	11
Drought	0	1	4	4	0	1	9
Disease	6	6	2	2	0	4	8
Native Predators	0	0	1	2	6	0	8
Reduction In Food Resources	0	2	0	1	1	1	7
Disturbance At Breeding Sites	1	1	0	0	6	0	7
Recreation	1	1	3	1	1	0	5
Exotic Inverts	1	1	0	0	0	0	2
Competition with Native spp	1	0	0	0	0	0	1
Total	**192**	**184**	**165**	**131**	**53**	**41**	**612**

5. Estimation of Costs

Methods

The Cost of Management

The cost of managing a threat depends, among other things, on how ambitious the objective is. For example, the cost of eradicating a weed is much larger than achieving strategic containment. For the costs to be comparable across threats, the management goals need to be comparable. We referred to the INRM plan to define the goal for each of the threats. So, for example the plan has resource condition targets:

> *RCT3-3 By 2020, there will be no decline in the conservation status of any 2005 listed threatened species or communities, and no additional species or ecological communities will require formal listing as threatened as a result of continued threatening processes.*
>
> *RCT3-4 By 2020, there is strategic containment of declared weeds, ecologically invasive plants and feral animals, sufficient to ensure that they have no significant impact on the conservation status of any Territory species or ecological community.*

Our task therefore was, in each region and vegetation type, to estimate the cost of ensuring that these targets are met.

To calculate the cost of management, we estimated the personnel and operational costs each year over a defined number of years. If the job will take many years, a discounting function was used to reduce the present cost of such future work. Where management is already occurring, this cost was calculated as well.

It was decided to undertake a cost-benefit analysis taking account of the costs involved in attaining the stated RCT, discounted for the future but expressed as net present value (meaning the cost is how much it would cost in today's terms, but accounting for the fact that delayed actions cost less). For some of the threats, mitigation results in increased income, for example from higher land productivity (e.g. control of *Mimosa*), whereas for others the opposite is true (reducing land clearing means a loss of potential production). These factors are all taken into account in the method.

The full analysis for each threat is described in the Appendices, but the method is illustrated here using Land Clearing as an example. An extended cost-benefit analysis was carried out with regard to land clearing activities. Discounting the stream of future costs and benefits over a given time horizon and for two different scenarios permits a net present value for each scenario to be calculated. A comparative analysis of the scenarios was subsequently carried out. In addition to the private (landholder financial) values that are associated with land clearing, public conservation values were accounted for in the form of carbon sequestration services.

The baseline scenario was one where there is no (binding) land clearing constraint. The alternative scenario was one where a land clearing constraint implies that a certain proportion of an individual property may not be cleared. Land clearing costs include a one-off expenditure for the clearing of the land *per se*, increased fire management costs associated with cleared land management and the costs associated with the loss of any income from carbon sequestration services.

Land clearing benefits include the gross margin of the agricultural activity being undertaken on the cleared land.

A sensitivity analysis was carried out on a range of variables, including the discount rate, time horizon, land clearing costs, the extent of the land clearing constraint and the value of carbon sequestration services.

Results

Cost calculations were compiled for management of feral animals, land clearing and fire. The overall costs - expressed as the net present cost over 20 years - are presented in Table 13. For this analysis, the feral herbivores are split into the Arnhem Land region and all others. For all others regions, feral herbivore control has a net economic benefit because the number of cattle that pastoralists can raise increases as the number of ferals is reduced. In the Arnhem region, there is no pastoral industry so there is a net cost. Feral predators are split into cats and foxes because the estimated cost of controlling cats is approximately 100 times greater than foxes. The net present cost ranking (1 being the most expensive) is compared with the threat ranking (1 being the largest threat across all regions). The investment priority column broadly reflects the ratio of these two rankings, expressed as high, medium and low return for investment.

Two of the threats have an economic benefit (feral herbivore (excluding Arnhem)) and clearing), so the investment priority is high as management of these should be undertaken irrespective of the level of threat. Of the others, the program that aims to prevent cane toad colonisation of islands is the cheapest and, together with feral herbivores in Arnhem Land, foxes and fire all have a medium investment priority ranking. Notice that fire management has a relatively high cost - over $400 m - but this may be offset to a considerable extent by the contribution of fire management to greenhouse gas abatement (as is already the case in the WALFA project in Arnhem Land) or to carbon sequestration. It is beyond the scope of this review to predict whether these contributions will entirely cancel the cost as calculated here. The implied investment priority for controlling rabbits, pigs and cats are much less appealing. These are all species with high costs (extreme in the case of cats) and relatively low threat ranking. Notice that for cats the enormous cost is based on excluding them from only 10% of the area they occur in, whereas for pigs and rabbits the cost is for controlling their entire populations.

Table 13. Net present costs, threat rankings and investment priorities for a range of threats.

Species	Net Present Cost over 20 years ($m)*	Net Present Cost Ranking[1]	Threat Ranking[2]	Investment Priority
Feral Herbivores (except Arnhem)	-186.55	9	1	High*
Land Clearing	-101.42	8	23	High*
Island Cane Toads	1.12	7	15	Medium-High
Feral Herbivores (Arnhem only)	5.82	6	6[3]	Medium
Feral Predators (Foxes only)	33.60	5	10[4]	Medium
Pigs	82.35	4	13	Low
Fire	444.00	3	7	Medium
Rabbits	498.92	2	11	Low
Feral Predators (Cats only)	5,277.30	1	10[4]	Low

* Negative net present costs represent benefits resulting from control (e.g. from reduced competition with cattle or as a result of accounting for CO_2 sequestration values. Given that benefits form control are generated, the associated investment priority is high.

[1] 1 being the most expensive

[2] 1 being the largest threat across all regions

[3] This ranking is for the Arnhem Land region only (the others are for the entire NT).

[4] Cats and foxes were considered together in the threat analysis, and so share a ranking here.

6. Conclusions

Since not all of the costs are currently available, the relative benefits of control cannot be compared comprehensively among all threats. However, several significant conclusions can be made.

1) Controlling threatening processes will be a huge financial challenge. Even ignoring the cost of feral cat control and without the weed and pastoralism costings, we estimate that more than $1 billion will be needed over the next 20 years to achieve the objectives of the INRM Plan.

2) The control of feral herbivores was found to be the highest priority. These species are already routinely controlled in several NT regions, and this action is well justified by the analysis here. Moreover, the economic benefits of feral herbivore control exceed the costs (except in Arnhem Land), so there is scope for expanding these programs without competing for scarce NRM funds, and possibly even of using the economic benefit in some regions to subsidise control in Arnhem Land.

3) Introduced pasture grasses are highly threatening and need to be managed. Currently, there is relatively little management of any of these species, and some are still being planted.

4) Progress toward low-impact pastoralism is a priority because pastoralism was ranked as a substantial threatening process across most regions.

5) Inappropriate fire regime is a pervasive threat, but active fire management to re-impose a more benign regime across the whole of the NT will be expensive.

6) Feral predators, especially cats, represent a major challenge. One of the problems is that the impact of cats on biodiversity is not well understood, especially in the northern part of the NT. The other problem is the very large cost of control for cats. It seems that unless a more cost effective control method can be found, there is little prospect of controlling cats on a broad scale. The situation with foxes is much clearer, because their impact is better understood and the costs of control are much less. A broad scale program, modelled on the Western Shield program in Western Australia, would cost about $33 m over 20 years.

7) Current control action targeting weeds is appropriate for most species (especially mimosa, parkinsonia, prickly acacia).

8) Several weed species are “sleepers”: they do not need large control efforts, but should be monitored and controlled vigorously if and where they occur (e.g. rubbervine, bellyache bush, Brazilian pepper).

9) Introduction of carbon pricing as a means of limiting climate change will change the cost-benefit ratio for land clearing and fire management to make them much more attractive. In the case of land clearing, even a modest price for carbon will mean that retaining native vegetation will be more economically productive than replacing it with pasture or broadacre cropping.

10) The impacts of climate change on biodiversity will be far-reaching and perhaps more serious than any of the threats reviewed here. Moreover, these threatening processes will change in unpredictable ways and may become more severe in the future. Each of the threatening processes makes it harder to protect biodiversity from climate change, so the need to control these threats becomes even more urgent when climate change is also factored into planning.

7. References

Anon (2005) *Integrated Natural Resource Management Plan for the Northern Territory*. Landcare Council of the NT, Darwin.

Ash AJ, Hunt LP, Petty S, Cowley R, Fisher A, MacDonald N and Stokes C (2006) Intensification of pastoral lands in northern Australia. In *14th Biennial Australian Rangelands Conference*. Renmark, SA. pp. 43–46. Australian Rangeland Society, Brisbane.

Biograze (2000) *Biograze: waterpoints and wildlife*. CSIRO, Alice Springs.

Blake G (2005) 'Object oriented mapping of sandstone heath vegetation on the Arnhem Plateau.' Unpublished BSc (Hons.) thesis. (Charles Darwin University: Darwin)

Bolton BL and Latz PK (1978) The western hare-wallaby, *Lagorchestes hirsutus* (Gould) (Macropodidae), in the Tanami Desert. *Australian Wildlife Research* **5**, 285-293.

Burbidge AA and McKenzie NL (1989) Patterns in the modern decline of Western Australia's vertebrate fauna: causes and conservation implications. *Biological Conservation* **50**, 143-98.

Ehrlich PR (1988) The loss of biodiversity. In: *Biodiversity* (ed E. O. Wilson) pp. 21-7. National Academy Press, Washington, DC.

Fairfax RJ and Fensham RJ (2000) The effect of exotic pasture development on floristic diversity in central Queensland, Australia. *Biological Conservation* **94**,11-21.

Fisher A (2001) *Biogeography and conservation of Mitchell grasslands in northern Australia*. PhD thesis, Faculty of Science & Information Technology, Charles Darwin University, Darwin.

Fisher A, Hunt L, James C, Landsberg J, Phelps D, Smyth A & Watson I (2004) *Review of total grazing pressure management issues and priorities for biodiversity conservation in rangelands: A resource to aid NRM planning*. Desert Knowledge CRC and Tropical Savannas CRC, Alice Springs.

Fisher A and Kutt A (2006) *Biodiversity and land condition in tropical savanna rangelands: summary report*. Tropical Savanna Management CRC, Darwin.

Fisher A and Kutt A (2007) *Biodiversity and land condition in tropical savanna rangelands: technical report*. Tropical Savanna Management CRC, Darwin.

Franklin DC (1999) Evidence of disarray amongst granivorous bird assemblages in the savannas of northern Australia, a region of sparse human settlement. *Biological Conservation* **90**, 53-68.

Grice AC (2006a) The impacts of invasive plant species on the biodiversity of Australian rangelands. *Rangelands Journal* **28**, 27-35.

Hennessy K, Page C, Bathols J, McInnes K, Pittock B, Suppiah R and Walsh K (2004) *Climate Change in the Northern Territory*. Climate Impact Group, CSIRO Atmospheric Research.

IPCC (2001) C*limate Change 2001: Synthesis Report*. Intergovernmental Panel on Climate Change.

IPCC (2007) *Climate Change 2007: Synthesis Report*. Intergovernmental Panel on Climate Change.

James CD, Landsberg J & Morton SR (1999). Provision of watering points in the Australian arid zone: a review of effects on biota. *Journal of Arid Environments* **41**, 87-121.

Landsberg J, James CD, Morton SR, Hobbs T, Stol J, Drew A & Tongway H (1997). *The effects of artificial sources of water on rangeland biodiversity*. Environment Australia and CSIRO, Canberra.

Landsberg J, O'Connor TG and Freudenberger D (1999). The impacts of livestock grazing on biodiversity in natural systems, In *Vth International Symposium on the nutrition of herbivores*(Eds, Jung, H.-J. G. and Fahey, G. C.) American Society of Animal Science, Savoy, Illinois.

Lowe I (2005) *The Big Fix: Radical solutions for Australia's environmental crisis*. Black Inc., Melbourne.

Ludwig JA, Tongway DJ, Freudenberger DO, Nobel JC & Hodgkinson KC (1997) *Landscape ecology, function and management: principles from Australia's rangelands.* CSIRO, Melbourne.

Martin TG and van Klinken RD (2006). Value for money? Investments in weed management in Australian rangelands. *Rangeland Journal* **28**, 63-75.

Noss RF (1991) *Landscape connectivity at different scales*. Island Press, Washington DC.

Oxley TJ, Leigo S, Hausler P, Bubb A, and Macdonald NM (2005) *Pastoral Industry Survey NT 2004*. Department of Primary Industry, Fisheries and Mines, Darwin.

Rankmore BR and Price OF (2004) The effects of habitat fragmentation on the vertebrate fauna of tropical woodlands, Northern Territory. In *Australian Forest Ecology (second edition)* (ed D. Lunney.), pp. 452-473. Royal Zoological Society of New South Wales, Mosman.

Russell-Smith J (1991) Classification, species richness and environmental relations of monsoon rain forest in northern Australia. *Journal of Vegetation Science* **2**, 259-278.

Smith NM (2002) *Weeds of the wet/dry tropics of Australia: a field guide*. Environment Centre, Darwin.

Thackway R and Creswell IDE (1995) *An interim biogeographic regionalisation of Australia: a framework for establishing the national system of reserves*. Australian Nature Conservation Agency, Canberra.

Thackway R and Lesslie R (2006) Reporting vegetation condition using the Vegetation Assets, States and Transitions (VAST) framework. *Ecological Management and Restoration* **7**, 53-62.

Williams J, Read C, Norton A, Dovers S, Burgman M, Proctor W and Anderson H (2001) *Biodiversity, Australia State of the Environment Report 2001 (Theme Report)*. CSIRO, on behalf of the Department of the Environment and Heritage, Canberra

Wilson BA, Brocklehurst PS, Clark MJ and Dickinson KJM (1990) *Vegetation Survey of the Northern Territory, Australia*. Conservation Commission of the Northern Territory.

Woinarski JCZ, Milne D and Wanganeen G (2001) Changes in mammal populations in relatively intact landscapes of Kakadu National Park, Northern Territory, Australia. *Austral Ecology* **26**, 360-370.

Woinarski JCZ and Dawson F (2001) Limitless lands and limited knowledge: coping with uncertainty and ignorance in northern Australia. In: *Ecology, uncertainty and policy: managing ecosystems for sustainability* (eds T. W. Norton, J. W. Handmer and S.R.Dovers), pp. 83-115. Longmans, London.

Woinarski JCZ and Fisher A (2003) Conservation and the maintenance of biodiversity in the rangelands. *The Rangeland Journal* **25**, 157-171.

Woinarski JCZ and Catterall CP (2005) Historical changes in the bird fauna at Coomooboolaroo, northeastern Australia, from the early years of pastoral settlement (1873) to 1999. *Biological Conservation* **116**, 379-401.

Woinarski J, Pavey C, Kerrigan R, Cowie I and Ward S (2007) *Lost from our landscape: threatened species of the Northern Territory*. Northern Territory Department of Natural Resources Environment and the Arts, Darwin.

Appendix A. Vegetation Clearing as a threatening process in the NT

1. Threat Scores

Calculation of clearing extent

Clearing extent is monitored and mapped annually by NRETA's Land Monitoring unit by interpretation of Landsat images from all of the regions where clearing is occurring. The minimum mappable cleared area is 1 ha. The 2005 data were intersected with the regions and vegetation types using a GIS, and the data summarised as the percentage of each vegetation type and region combination that had been cleared (Table A1).

Table A1. Percentage of each vegetation type and region combination that has been cleared (as of 2005). '-' indicates that the combination does not exist: all of the existing combinations have total areas of at least 200 km^2.

Vegetation type	Arnhem Land	Savanna	Barkly	Southern	Arid
Mangroves and Coastal	0.21	0.59	-	-	-
Rainforest & Riparian	0	1.33	-	-	-
Floodplain	0	1.1	-	-	-
Melaleuca	0.01	1.11	-	-	-
Seasonally inundated woodlands	0.01	2.51	-	-	-
Euc. forest	0.38	3.12	-	-	-
Euc. woodlands	0	0.08	0	0.24	0.03
Heath	-	2.96	0	-	-
Acacia	4.95	0	0	0	0
Grassland	-	0.01	0	0	0
Spinifex	-	0	0	0.05	0
ALL	0.27	1.08	0	0.03	0.01

Developing the threats table

The percentage clearing in Table A1 can be translated directly as the extent values in each of the attribute tables. That is to say, all five of the risk tables will have the same values for extent. For the assessment of threats, we assume that the total extent of clearing will double by the year 2030 (2030 being the forecast range for this project, to comply with Resource Condition Targets in the Integrated Natural Resource Management Plan). A doubling is consistent with the NT Cattlemen's Association' vision to double cattle production in the NT and with recent trends in the expansion of clearing for horticulture and other developments in the Daly, Tiwi, Ti Tree and Darwin regions.

Severity describes the actual site impact that clearing has on the five landscape attributes used in the project. When clearing is extensive, it can have an impact on landscape health beyond the clearing itself, into the uncleared land. For example, many ecological effects from fragmenting vegetation into small blocks have been documented, including problems with small population sizes and edge effects (wind throw, weed incursion, predator invasion).

However, the extent of clearing in the NT is currently so low (i.e. all below 5%), that there will be negligible fragmentation effects, at least at the scale used here. Generally, fragmentation only has an evident effect when clearing exceeds 70% of a landscape (Andren 1994; With & Crist 1995). For this reason, the severity of the impact on vegetation and species are considered only to result to the clearing itself and not to the retained vegetation. Similar arguments apply to the other attributes (landscape function, production and cultural value).

The severity was considered as follows for the five attributes:

Vegetation condition - clearing transforms the native vegetation and so severity is classed as VAST class 5. So for all vegetation type/region combinations where clearing has occurred, the severity is classed as 5.

Species - most native species will be lost from cleared areas but some will remain while a few 'weedy' species will benefit from the clearing (Rankmore & Price 2004). The severity class in all cases is 4.

Landscape Function - it is difficult to categorise the severity of impact to landscape function from clearing because this depends to a large degree on the land management. Historical experience in other parts of Australia is that clearing may contribute to soil erosion, loss of structure and function, and associated physical problems such as salinity and water logging (Anon 2001, Ash *et al*. 1992). However, methods are available to prevent soil erosion and preserve landscape function in pasture or cropping land. Clearing can have a major impact on aquatic ecosystems via reduction in flows through extraction (Erskine *et al*. 2003) or via unwanted discharges such as soil or chemicals (Anon, 2001). The extent to which all of these will occur is hard to predict, so a moderate severity has been used here (class 2).

Production - since clearing is done to increase production, we assume that there is no negative impact from this factor on production (class 0).

Cultural values – these differ among people, and while some people would consider that the spiritual, aesthetic and recreational value of land is completely lost when the vegetation is cleared, others would have the opposite view, considering these values to be enhanced by clearing. There is no information with which to determine where the balance lies among the people of the NT. However, given that the physical and ecological impact is great, we have assumed that there is a moderate impact on cultural values (class 2).

Implications of the threat values

Judged for the NT as a whole or for the regions considered here, land clearing represents a very small threat to biodiversity. The Savanna region had the highest area and percentage of clearing, but even this comprised only 1.1% of its area. Except for highly localised species in particularly targeted environments, it is unlikely that land clearing on its own will cause species to become threatened or for threatened species to become extinct in the forthcoming 20 years.

The impact of land clearing is related to scale. The Daly Basin is the most heavily cleared bioregion, with 10.9% of its area cleared, which is 10 times that for the Savanna region as a whole.

Clearing is an important contributor to greenhouse gasses. Currently, 601 000 ha of vegetation has been cleared. Assuming a carbon dioxide storage of 47 tonnes per ha stored in trees (data for the savannas, Chen *et al*, 2003), the net carbon emission is 28 million tonnes, or 141 tonnes per Territorian. This loss may be in part redressed by carbon storage in the plants that are grown in the cleared areas, but where the land use will be for intensive pastoral production, the greenhouse gas emissions will be much higher again because cattle produce large quantities of methane, which is 20 times more active as a greenhouse agent than carbon dioxide.

2. Cost-benefit Analysis for Land Clearing

Land clearing for agricultural purposes in the savanna lands of the NT results in environmental modification of and net loss of biodiversity on those cleared lands. There is a divergence between the private (financial) benefits of land clearing for agricultural production (benefits accruing to the landholder) and the cost associated with biodiversity loss arising from that land clearing (costs accruing to society as a whole). Society thus faces a trade-off between agricultural production and biodiversity conservation.

In the absence of government intervention/regulation, it may be expected that landholders will treat the loss of biodiversity as an externality, focusing instead only on the net production benefits of land clearing for agricultural production. Under such circumstances, where the benefits from agricultural production outweigh the costs of clearing land, it may be expected that landholders will clear all the arable land available.

Gross margin of agricultural production on cleared land

The Douglas-Daly region is one of the few areas in the NT where land of agricultural potential is found in the vicinity of adequate water supplies. As a result, agricultural production is potentially profitable. Based on figures from DBERD (1999, unpublished), gross margins for arable land categories A&B are $550 per ha for irrigated maize and $390 for dryland cavalcade p.a. (in 1999 Australian dollars). The calculation of the gross margins includes such factors as pre-harvest, harvest and post-harvest costs. We assume water limitations mean production on these lands is split equally between the two, resulting in a mean gross margin of $470/ha p.a. For arable land categories BW, C and CW, which are most typically used for finishing steers, gross margin is approximately $170/ha p.a. Given the roughly equal proportion of these two overall land types, the average gross margin of agricultural land in the Daly-Douglas region is (470+ 170)/2 = $320 per ha per year or, adjusted for changes in the CPI, $400/ha p.a. in 2006 dollars (ABS, 2006).

Costs of land clearance and fire management

The above agricultural gross margin per hectare values are for land that has already been cleared. The one-off costs of land clearing must also be considered in order to properly determine the private profitability of agricultural production. Land is typically cleared by stretching a chain between two bulldozers moving in parallel and burning the resulting felled vegetation. Clearing activities are usually contracted out and cost $750 per hectare (personal communication from contractors).

In addition, cleared agricultural land requires a changed fire and weed management regime compared to uncleared land. The additional cost of managing cleared land, which involves periodic prescribed fires and fighting of wildfire, is taken here as $0.71/ha p.a. (cost from the fire section of this report). (Note that this assumes that this expenditure reduces the risk of crop or livestock loss due to fire to zero)

Rate of land clearance and proportion of arable land

The mean rate of clearing (actually cleared land plus land licensed to be cleared but not actually cleared due to the imposition of a clearing moratorium) over two years in the Stray Creek Blocks is assumed to be representative of the average rate of land clearing in the Douglas-Daly region in the absence of government regulation (beyond pre-existing land clearing regulations that stipulate buffers on watercourses etc). This is equal to a rate of 3,912 ha p.a.

All of this land may be considered to be arable land (otherwise it would not have been, or planned to have been, cleared) and, based on a sample of 18 properties, such arable land represents an average of 71.4% of the land on individual properties (arable land calculated from Land Unit mapping: Aldrick & Robinson 1972).

Net present value of agricultural production in the absence of a land clearing constraint

The net present value (NPV) of agricultural production can now be calculated. Given the temporal nature of land clearing costs (a one-off cost at the beginning of the time horizon) and the flow of agricultural benefits across the time horizon under consideration, it is necessary to use discount rates so that the different costs and benefits at different points in time may be compared. For the purposes of this analysis a 20 year time horizon is chosen and a 5% discount rate is used. The 20 year time horizon was used to align the analysis with the Resource Condition Targets under the INRM planning framework. Both assumptions are subject to a sensitivity analysis later in this paper.

Based on 3,912 ha being cleared per year during 20 years (totalling 78,240 hectares of arable land cleared), each with a one-off clearing cost of $750/ha and a gross agricultural margin of $400/ha p.a. in each of the 20 years under consideration, and using a 5% discount rate, the NPV of unconstrained land clearing is equal to $133.81 million. This is equivalent to an annualized NPV of $10.74 million p.a.

Net present value of agricultural production in the presence of a 50% land clearing constraint

There is currently a moratorium on land clearing in the Daly-Douglas region. Based on the above calculations this may be considered to be costing the farmers a total of $133.81 million or the equivalent of $10.74 million p.a. This is of course a private measure of the opportunity cost from agricultural income forgone and excludes the public benefits to society in general of the biodiversity conserved.

In the case that the current moratorium on land clearing may be lifted, such private losses will be reduced. Assuming a regulation is put into place that instead restricts land clearing on individual properties to 50%, then 1,173 ha p.a. (calculated for the sample of 18 properties based on the area of arable land that exceeds 50% of the total area) out of the 3,912 ha p.a. of arable land will not be allowed to be cleared and only 2,739 ha of arable land will be allowed to be cleared. This is equivalent to not clearing 23,450 ha (1,173 x 20) of arable land over 20 years.

In the presence of a 50% restriction, landholders would only be able to earn $93.70 million (54,790 ha cleared), whereas in the absence of any regulation landholders were able to earn $133.81 million (78,240 ha cleared). Hence, the private opportunity cost to landholders of the agricultural income forgone is $40.11 million over 20 years or equivalent to $3.22 million p.a.

Present value of CO2 sequestration services of uncleared land

As previously noted, $40.11 million is the private (financial) opportunity cost incurred by landholders as a result of a 50% land clearing regulation. This value, however, ignores the benefits that accrue to society in general from having conserved biodiversity and environmental services on 23,450 ha of land that would otherwise have been cleared.

While a number of techniques for valuing such biodiversity exist, valuing such biodiversity will be complex and relatively expensive given the many non-market values that are likely to be involved. Before embarking on such an endeavour, it is worth considering the incorporation of other, easier to quantify, public values into our cost-benefit analysis of land clearing and agricultural production in the Douglas-Daly region.

The most obvious of these are the carbon sequestration services provided by uncleared land. Relative to cleared land converted to pasture, uncleared land sequesters an additional 193 tons/ha of CO_2 (NT Greenhouse Office working value for savannas; J. McCallister pers. comm.). This analysis depends on the price of carbon, which is hard to predict since no carbon trading system is in place for Australia. We have assumed the price will be $25 per ton of CO_2. This is toward the lower end of the ranges used in most reviews of the likely implications of carbon trading (e.g. IPCC 2001, Hatfield-Dodds *et al.* 2007). Combining the values for the carbon biomass in woodlands and the carbon price, every hectare of uncleared land provides $4,825 of CO2 sequestration services.

Where no land clearing restriction is enforced, we noted that 3,912 hectares p.a. would be cleared. This is equivalent to $18.86 million p.a. in lost CO2 sequestration services or $235.23 million over 20 years.

Under the 50% restriction, 2,739 (3,912-1,173) hectares p.a. would be cleared. This is equivalent to $164.73 million over 20 years in lost CO2 sequestration services or the equivalent of $13.22 million p.a.

How such values would play a role in affecting land clearing decisions depends on how the initial property rights are defined. Currently, the actual value of the CO2 sequestration services of the land is zero as no market for the CO2 sequestration services has been created in this part of the NT. Hence, landholders treat the carbon sequestration services of their land as an externality and society pays the cost of increased greenhouse gas emissions.

However, if landholders were to be paid for the CO2 that their land continued to sequester or, alternatively, if they had to pay for lost CO2 that their land clearing activities caused, then they would rapidly internalise this externality into their financial decision-making framework regarding whether to clear land or not.

Net present value of arable land in the presence of CO2 emissions charges

As can be seen from the above CO2 sequestration value calculations, it is apparent that such values in fact swamp any benefits that can be obtained from agricultural production.

In the absence of any land clearing restrictions but under a system where landholders either received a payment for maintaining CO_2 sequestration services or were charged for their destruction, then landholders would find it more profitable to not clear any land. Under a CO_2 payment scheme (similar to the ConocoPhillips carbon offset payments in Arnhem Land) landholders could earn $235.23 million over 20 years for not clearing land compared to the $133.81 million they could have made from clearing the land for agriculture (a benefit to landholders of $101.42 million).

Alternatively, if landholders were obliged to pay for loss of the CO_2 sequestration services arising from land clearing, the economically wisest course for them would be to not clear any land as clearing would otherwise result in a loss of $101.42 million ($133.81-$253.23) over 20 years.

Under the 50% land clearing restriction scenario landholders would forgo $164.73 ($70.50-$235.23) million from cleared land while earning only $93.70 million over 20 years from agricultural production. Hence, no land clearing would be more profitable under this scenario too.

Note also that since these CO_2 sequestration values swamp the returns to agricultural production, adding in the value of biodiversity per se will not change our overall findings i.e. agricultural production is not maximising economic returns – from a public perspective, although we should continue to treat these values as lower bound estimates since we have only considered a single environmental service.

Sensitivity analysis

Adjusting the time horizon from 20 years to a shorter or longer time horizon changes the net present value over the time horizon under consideration but does not change the overall findings. Similarly, changing the discount rate does not make a significant difference to the overall result (a higher rate reduces the value of future benefits, while a lower rate increases their value, relatively speaking).

Changes in the cost of land clearing also do not play a significant role, although a lower cost would make clearing for more intensive agriculture somewhat more attractive.

Changes in the restriction regulation also affect NPV (lower restriction rates generate lower agricultural opportunity costs for landholders in the absence of CO_2 pricing) but are bounded by the figure of $133.81 million (landholders make this amount in the absence of any restriction and forgo this amount where a 100% restriction is in place.

The rate of land clearing affects the opportunity cost imposed on producers if a cap is placed on farming, as realised over the 20 year time frame of this analysis (in the absence of a CO_2 emission charges, see below). Put simply, the faster the clearing is, the more producers will lose from imposing limits on clearing. However, there will be a finite limit on the total amount of clearing that can occur in the NT due to the distribution of suitable land and in any case the opportunity cost per ha of clearing foregone remains the same irrespective of the proposed level of land clearing. In other words, if the government were to compensate producers for not clearing land they otherwise would have, then the total amount of compensation depends on the rate at which producers would have cleared in the absence of controls, but the compensation rate per ha of this foregone clearing is constant. CO_2

emissions pricing reverses the opportunity loss to a benefit from not clearing so the argument above would be reversed (the absolute benefit depends on the rate of clearing foregone but the per ha benefit is constant).

Given that the CO2 sequestration values tend to swamp the agricultural values, it is worth exploring at what rate the economic valuation CO2 would need to be in order for this not to happen. An iterative approach reveals that at any price above $14/ton carbon sequestration services will be more valuable than agricultural production. Similarly, the sequestration is only more valuable than agricultural production as long as the carbon storage in the cleared woodland is at least 108 tonnes/ha higher than the vegetation that replaces is (with a $25 carbon price).

Discussion of the costs results

Note that the above model provides a static analysis. Payment for CO2 sequestration services may promote dynamic changes to production systems that retain sequestration services while permitting economic production (e.g. through agroforestry production). In such a case, the above calculations regarding the relative benefits of CO2 sequestration service maintenance over economic production will tend to be over-stated over the longer-term.

More seriously, in the absence of CO2 values, the costs of adopting a command and control (C&C) approach, in terms of imposing a blanket percentage of land that may not be cleared restriction is likely to be high compared to more sophisticated market-based instrument approaches. This is because such a C&C approach ignores that fact that not all arable land types are equally profitable, not every bit of arable land is equally important in terms of biodiversity conservation, and larger conserved areas with corridors are likely to be more effective at conserving biodiversity than smaller unconnected ones on each individual property.

3. References

Aldrick JM and Robinson CS (1972) *Report on the Land Units of the Katherine-Douglas area, NT, 1970*. Animal Industry and Agriculture Branch: Northern Territory Administration, Darwin.

Andren H (1994). Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos* **71**, 355-366.

Anon (2001) *Australian Agriculture Assesment 2001, volume 1*. National Land and Water Resources Audit, Canberra.

Ash AJ, McIvor JG and Winter WH (1992) Managing rangelands for production without degradation. In *Conservation and development issues in northern Australia*. (eds I. Moffat and. A. Webb.), pp 171-180. Australian National University, Darwin.

Chen X, Hutley LB and Eamus D (2003) Carbon balance of a tropical savanna of northern Australia. *Oecologia* **137**, 405-416.

Ehrlich PR (1988) The loss of biodiversity. In: *Biodiversity* (ed E. O. Wilson) pp. 21-7. National Academy Press, Washington, DC.

Erskine WD, Begg GW, Jolly P, Georges A, O'Grady A, Eamus D, Rea N, Townsend S, and Padovan A (2003) *Recommended Environmental Streamflows for the Daly River, Northern Territory, based on Ecological, Hydrological and Biological Principles*. Supervising Scientist, Darwin

Hatfield-Dodds S, Carwardine J, Dunlop M, Graham P and Klein C (2007) *Rural Australia Providing Climate Solutions: preliminary report to the Agricultural Alliance on Climate Change*. CSIRO Sustainable Ecosystems, Canberra.

IPCC (2001) *Climate Change 2001: Synthesis Report*. Intergovernmental Panel on Climate Change.

Noss RF (1991) *Landscape connectivity at different scales*. Island Press, Washington DC.

Rankmore BR and Price OF (2004) The effects of habitat fragmentation on the vertebrate fauna of tropical woodlands, Northern Territory. In *Australian Forest Ecology (second edition)* (ed D. Lunney.). pp. 452-473. Royal Zoological Society of New South Wales, Mossman.

Williams J, Read C, Norton A, Dovers S, Burgman M, Proctor W and Anderson H (2001) *Biodiversity, Australia State of the Environment Report 2001 (Theme Report)*. CSIRO on behalf of the Department of the Environment and Heritage, Canberra.

With KA (1997). The Application of Neutral Landscape Models in Conservation Biology. *Conservation Biology* **11**, 1069-1080.

Appendix B. Fire as a threatening process in the Northern Territory

1. Threat Scores

Introduction

A number of data sources are available which make it relatively easy to provide landscape-scale assessments of the current extent, seasonality and frequency of fire in the Northern Territory (NT), and to make informed assessments of the impacts of contemporary fire regimes. Our understanding of fire occurrence in the NT has developed rapidly over the past two decades with the application of different satellite sensors (especially AVHRR, MODIS, LANDSAT) for fire mapping and detection ('hot spot') purposes, and associated development of ready public access to fire mapping information through development of dedicated websites (e.g. http://www.firewatch.dli.wa.gov.au; http:// www.firenorth.org.au). For example, fire mapping over the period 1997-2005 from daily, coarse-resolution (pixel size ~1 km^2) AVHRR imagery, clearly illustrates that, annually, most burning occurs in certain regions of the tropical savannas (Fig.B1).

Additionally, a large number of more regionally-focused studies have been undertaken in different habitats of the NT which have (1) described components of contemporary fire regimes, particularly with reference to finer-scale LANDSAT imagery (pixel size < 1 ha), (2) provided detailed conservation assessments of the impacts / effects of different fire regimes on biodiversity and production values. Many of these studies are summarised in a number of key reference materials: Myers *et al*. (2004), for the rangelands in general; Dyer *et al*. (2001), Williams *et al*. (2002) and Russell-Smith *et al*. (2003), for the tropical savannas region; and Allan & Southgate (2002), for the central Australian region. The fire and rangelands report (Myers *et al*. 2004) provides a particularly good source of information concerning the fire impacts on and status of different rangelands vegetation types.

In general terms, the above information sources indicate that, where fire does occur in the NT landscape today, it is mostly unmanaged, tends to occur as extensive, relatively intense wildfire under severe fire-weather conditions, either late in the dry season (Aug-Nov) in the savannas, or in spring-summer in central Australia. Impacts of these contemporary fire regimes vary greatly, differing with respect to types of habitat and the fire-response traits of individual species. For plants, for example, species possessing the capacity to resprout following burning (i.e. resprouters, like all eucalypts) are at a significant advantage in situations with frequent fire, compared with species which regenerate only from seed sources (i.e. obligate seeders, like many acacia shrubs) when adult plants are killed.

The studies referenced above indicate that, currently in the NT, the following habitat types are particularly at risk from frequent burning - rainforests, heathlands and acacia shrublands, stands of the long-lived obligate seeder conifers, *Callitris glaucophylla* (desert cypress-pine) and *C. intratropica* (northern cypress-pine). Contemporary fire regimes are also implicated in the demise of mammals (e.g. Bolton & Latz 1978, Woinarski *et al*. 2001) and granivorous birds (Franklin 1999). The spread of flammable introduced pasture grasses (e.g. gamba grass in northern savannas, buffel grass in central Australia) is likely to increasingly exacerbate problems associated with intense frequent fires.

Utilising the above references, the main fire responses of the 10 major vegetation / habitat types being addressed in this assessment are summarised in Table B1

Developing the fire threats table

The fire threats table developed for this assessment was stored in an excel file with 6 worksheets as follows:

- "Fire" worksheet—this is an additional column which provides background fire extent and severity information which has been used as the basis for making the threats assessments in other worksheets. For each of the 10 major vegetation / habitat types, this worksheet provides:
    (a) the mapped proportion occupied by each vegetation type in each of 5 "grazing land management zones"—GLMZ (Arnhem Land, Savanna, Barkly, Southern, Arid), with reference to a colour-coded scale (0%, <1%, 1-5%, 5-10%, >10%)
    (b) in the "Extent" column—the mean annual proportion of each vegetation type in each GLMZ which has been burnt over the period 1997-2005, derived from mapping of large fires from AVHRR imagery
    (c) in the "Severity" column—the proportion of the mean annual 'Extent' of each vegetation type in each GLMZ which has been burnt in the late dry season period (i.e. after June), using the same fire mapping data.

- Assessment worksheets—provides assessments of threats to respective values (Vegetation condition, Sensitive and threatened species, Landscape function, Production, Cultural), with respect to "Extent" of burning (derived from "Fire" worksheet, and expressed as classes as follows: 0 = <5% burnt; 1 = 5-10%, 2 = 10-25%, 3 = 25-50%), using the "Severity" classes as designated for respective values. Colour coding scale for the proportion of each vegetation type in each GLMZ is also as given in "Fire" worksheet. "Severity" is scored only for vegetation types where these occur in the GLMZ.

Threat assessment

Vegetation condition—generally speaking, all vegetation types that scored 2 or more with respect to "Extent" (i.e. mean annual extent of burning >10%) were assessed as being "significantly modified" by current fire regimes. Overall, fire impacts were assessed as being significantly greater in Arnhem Land, Savanna, and Southern GLMZ's. Thus 7 of a total of 8 vegetation types occurring in the Arnhem Land GLMZ, 8 of 10 in the Savanna GLMZ, and 3 of 4 in the Southern GLMZ, were assessed as being "significantly modified" by contemporary fire regimes. Fire impacts on all other vegetation types occurring in respective GLMZ's were assessed as "slightly modified".

Sensitive and threatened species—there is evidence of population declines among many species in a variety of vegetation types in the NT as a result of changed fire regimes. In particular, direct evidence of the decline in *Callitris intratropica* is generally taken as an indicator of declines in other species (Bowman & Panton 1993), and circumstantial evidence suggest that there is a range of plant species which need at least five years between fires and that this occurs relatively rarely in the savannas (Russell-Smith *et al.* 2003). However, there is no evidence that contemporary fire regimes have led to the regional extinction of any species.

Landscape function—given the general relationship between "vegetation condition" and "landscape function", especially with respect to habitat values, this assessment resembled closely that of 'vegetation condition', with most vegetation types that were scored two or more with respect to "Extent" considered to be significantly modified. Additionally, small areas of heath, acacia, and spinifex vegetation types in the Barkly GLMZ were also scored as "significantly modified" given that relatively high fire frequencies are likely to have affected their habitat amenity.

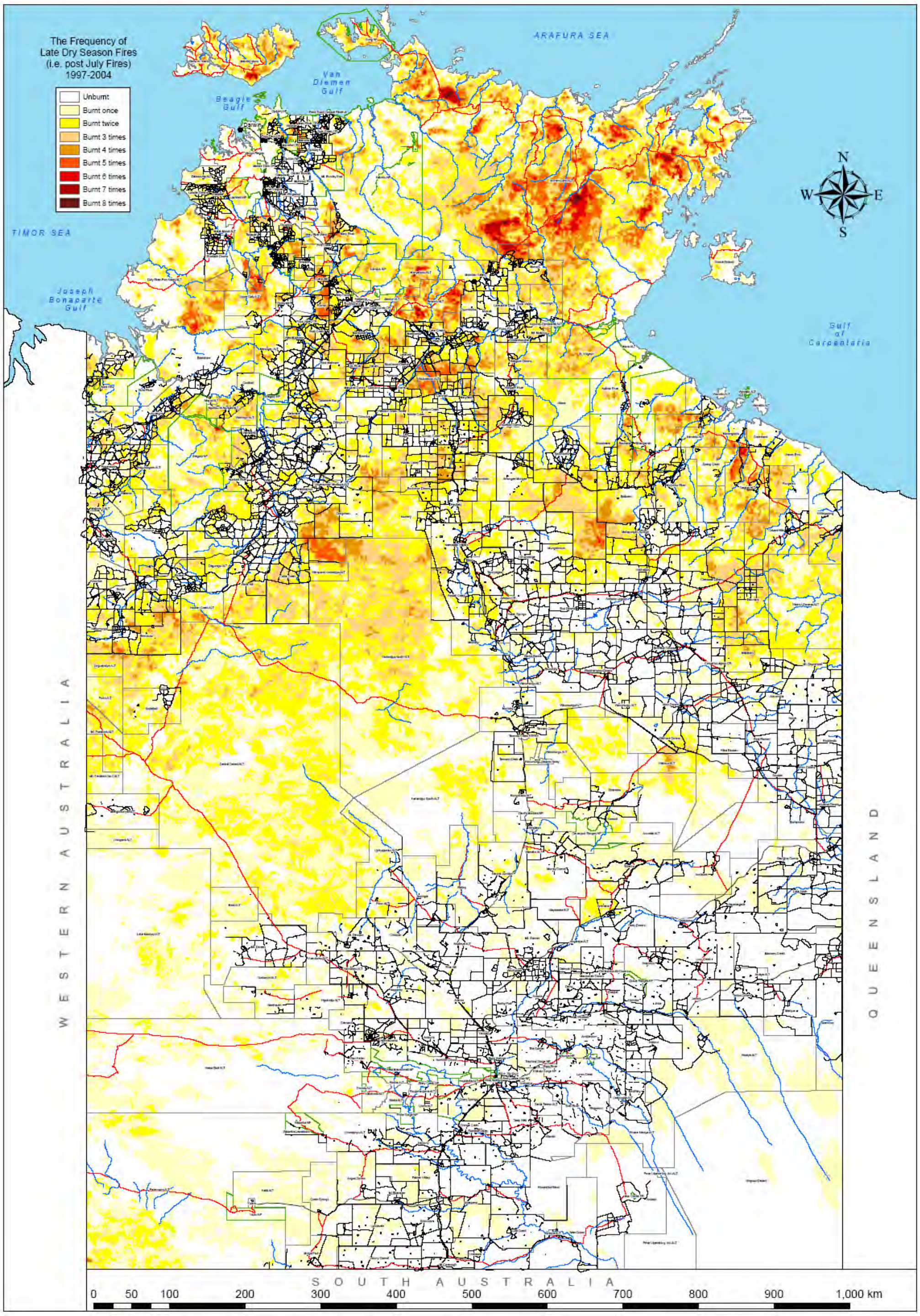
Production—detrimental impacts of fire on production values were considered to be restricted to 'tussock grassland' and 'Spinifex' vegetation types in the GLMZs in which they occur. Low fire frequencies in 'tussock grassland' in the Barkly, Southern and Arid GLMZs were assessed as having potential to reduce 'gross production by 2-10%' over the longer term given woody thickening issues. Conversely, relatively frequent fire in 'Spinifex' may lead to the degradation of this resource in the longer term.

Culture—in general, we gave a score of at least '1' ("obvious but minor") for all vegetation types occurring in respective GLMZs in recognition of changed cultural values associated with burning / not burning, associated with the shift from traditional / customary Aboriginal fire management to contemporary practices. We scored '2' ("up to 1/3 of value gone") for frequently burnt 'rainforest / riparian' and 'heath' vegetation types, in recognition of contemporary cultural values concerning their conservation significance.

Table B1: Summary fire responses of vegetation types addressed in this assessment

Vegetation type	Fire responses	References
Mangrove / coastal	not applicable	--
Rainforest / riparian	intense fires, particularly in late dry season, have major impacts on forest margins, and entry of flammable grasses	Russell-Smith & Bowman 1991; Russell-Smith & Stanton 2002
Melaleuca	generally resilient to recurring fire, except in floodplain situations with peaty soils (refer above). In absence of burning / overgrazing, Melaleuca invasion can occur rapidly, resulting in woody thickening and loss of open grassland habitat conditions.	Myers *et al*. (2004)
Floodplain / seasonally inundated	very dynamic habitat; herbaceous vegetation typically highly resilient to burning, except in places with peaty accumulation, and where smouldering fires consume plant root matter	Whitehead & McGuffog 1997; Myers *et al*. (2004)
Eucalypt forest	generally resilient to recurring fire, although intense fires impact on fire-sensitive elements (e.g. *Callitris*). Maintaining a fine-scale fire mosaic is important for granivorous birds and mammals with small home ranges.	Williams *et al*. 2002; Myers *et al*. (2004)
Other Eucalypt woodland	as for Eucalypt forest	Williams *et al*. 2002; Myers *et al*. (2004)
Heath	contains many obligate seeder shrub species and thus particularly sensitive to short intervals between fires which don't allow for production of new seed crops. Fire patchiness required to facilitate seed dispersal onto burnt sites.	Russell-Smith *et al*. 1998, 2001
Acacia shrublands	includes extensive areas of long-lived obligate seeder trees, especially mulga (*A. aneura*) and lancewood (*A. shirleyi*), as well many obligate seeder shrubs in heathy habitats. Frequent fires affect all these species as discussed for heaths above.	Allan & Southgate (2002); Myers *et al*. (2004)
Tussock grassland	comprises the pastorally important Mitchell grasslands which today are seldom burnt. This, combined with heavy cattle grazing pressure, can lead to woody thickening. Intense fires are then required for woody plant control. Patchy fires can be used to promote more sustainable pasture management by evening out grazing pressure, and rejuvenating moribund pastures	Myers *et al*. (2004)
Spinifex (hummock) grassland	very extensive habitat in central Australia, and also rocky sandstone ranges in north. Rapid build-up of Spinifex fuels (associated with rainy conditions in the Centre) creates very large fuel loads over extensive areas. Under such conditions, fires in Spinifex communities are typically intense and extensive, to the detriment of more fire-sensitive biota..	Allan & Southgate (2002); Myers *et al*. (2004)

Figure B. Fire frequency in NT, 1997-2005, mapped from AVHRR imagery (source: WA Dept Land Information)



2. The costs of fire management in the NT

Background

This section presents one approach to costing the objective of achieving an ecologically desirable fire management regime across the NT that is compatible with the stated goal of the INRM Plan. It assumes that a management regime of the type exemplified by the West Arnhem Land Fire Abatement (WALFA) program could achieve such a goal if extended across the entire Territory.

WALFA[1] involves the implementation of strategic fire management by project partners from early in the dry season in order to reduce the size and extent of unmanaged wildfires.

WALFA fire management costs are $71/km^2 p.a. (J. Russell-Smith, *pers. comm.*). However, Western Arnhem Land is a biodiverse area requiring a high level of expenditure on fire management and is therefore not necessarily representative of the costs for fire management on all land types in the NT. In other areas of Arnhem Land and the savannas in general, management costs are lower. This is because in the semi-arid and arid parts of the NT wildfires are naturally less frequent.

Cost Calculations

For the purposes of this project, we assume that all conservation reserves in the Savanna and Arnhem regions, plus the Tiwi Islands have high levels of biodiversity and hence do indeed require $71/km^2 p.a in management costs. These areas cover 60,173 km^2 or 4.5% of the total 1.35m km^2 under consideration.

However, we assume that all other parts of the Arnhem and Savanna regions (464,480 km^2 comprising 34.4% of the total land area) require only half of this cost ($35.5/km^2 p.a.).

The remaining Arid, Southern and Barkly regions (825,760 km^2 comprising 61.1% of the total land area) require roughly only half of that again ($18/km^2 p.a.) (J. Russell-Smith, *pers. comm.*).

Results

Accounting for the areas of each of these regions and their vegetation types, it is possible to assess the total costs for fire management to target levels across the NT[2].

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[1] The WALFA program is the result of a partnership between Darwin Liquefied Natural Gas (DLNG), the Northern Territory Government, the Northern Land Council and relevant Aboriginal Traditional Owners and indigenous representative organisations, formed to implement strategic fire management across 28,000 km^2 of Western Arnhem Land for the purposes of offsetting some of the greenhouse gas emissions from the Liquefied Natural Gas plant at Wickham Point in Darwin Harbour. The project aims to reduce greenhouse gas emissions from this area by adopting effective fire management practices in what is today mostly unmanaged land. Such practices will also help conserve environmental and cultural values in the project region equivalent to the adjacent World Heritage-listed Kakadu National Park. (Tropical Savannas CRC 2007) http://savanna.ntu.edu.au/information/arnhem_fire_project.html

[2] Full details contained in the fire management cost spreadsheet, available from the authors upon request.

The total annual cost is \$35.6m or an average of \$26.4 per km^2. This is split across high (\$4.3m), medium (\$16.5m) and low (\$14.9m) biodiversity areas. As can be seen in Table B2, annual fire management costs are highest in the savanna region, comprising 46.9% of the total.

Table B2. Annual Fire Management Cost (\$) by Region

Region	Annual fire management cost (\$)	
Arid	\$9.9m	27.8%
Arnhem Land	\$4.1m	11.4%
Barkly	\$1.7m	4.8%
Savanna	\$16.7m	46.9%
Southern	\$3.3m	9.2%
Total	\$35.6m	100%

Assuming a 20 year time horizon and a 5% discount rate, the total present costs of such a fire management program would be approximately \$444m.

Sensitivity analysis

The present cost and annual cost estimates are entirely dependent on the data provided by the relevant experts in this field. As described above, the model is driven by and is sensitive to the estimated fire management costs per management area and the land area considered to fall within each such management area.

With regard to the fire management costs per management area, the results are proportionally influenced by the costs (e.g. a doubling of these costs results in a doubling of the total present and annual costs). In addition, given the distribution of costs and land types, the results are most sensitive to a change in fire management costs in the medium biodiversity areas. As shown in Table B3, a doubling of those costs (while leaving the others unchanged) would increase annual (as well as total present) costs by 46.4%. Doubling of costs in low biodiversity areas would result in an increase of 40.6%, while in high biodiversity areas the difference would be only 12%. The results are therefore particularly sensitive to the estimated management costs in the medium and low biodiversity areas. This is so regardless of the magnitude or direction of the change in cost estimates.

Table B3. Annual costs under a doubling of costs in one given area

	Annual cost (\$) under a doubling of costs in one given area	Annual cost (\$) based on initial estimates	% increase relative to initial estimate
High biodiversity areas	\$39.9m	\$35.6m	12%
Medium biodiversity areas	\$52.1m	\$35.6m	46.3%
Low biodiversity areas	\$50.1m	\$35.6m	40.6%

Although annual costs are unaffected, total present cost results are sensitive to both the time horizon and discount factor used. Some indication of the maximum and minimum values that the total present costs may take are presented in Tables B4 and B5.

Table B4. Total Present Cost by discount rate over 20 year time horizon

Discount Rate (%)	0%	3%	5%	10%	15%
Total Present Cost ($)	$712.5m	$530.0m	$444.0m	$303.3m	$222.0m

Table B5. Total Present Cost by time horizon at 5% discount rate

Time Horizon (years)	10	20	50	Infinite
Total Present Cost ($)	$275.1m	$444.0m	$650.4m	$712.5m

Finally it is worth considering the issue of whether the total fire management costs are worth incurring. This can only be determined through an extended cost-benefit analysis, where the considerable benefits of fire management to, for example, biodiversity, carbon sequestration and cultural values would be taken into account. Although such an analysis is beyond the scope of this study, those values are likely to be considerable and may well exceed the average cost of $26.4/km^2 incurred by the above fire management program.

3. References

Allan GE and Southgate IR (2002) Fire regimes in the spinifex landscapes of Australia. In: *Flammable Australia: the fire regimes and biodiversity of a continent* (Eds RA Bradstock, JE Williams and AM Gill), pp. 145-176. Cambridge University Press, Cambridge.

Bolton BL and Latz PK (1978) The western hare-wallaby, *Lagorchestes hirsutus* (Gould) (Macropodidae), in the Tanami Desert. *Australian Wildlife Research* **5**, 285-293.

Bowman DMJS and Panton WJ (1993) Decline of *Callitris intratropica* R.T. Baker and H.G. Smith in the Northern Territory: Implications for pre- and post- European colonization fire regimes. *Journal of Biogeogography* **20**, 373-81.

Dyer R, Jacklyn P, Partridge I, Russell-Smith J and Williams RJ. 2001. *Savanna Burning: Understanding and Using Fire in Northern Australia*. Tropical Savannas Cooperative Research Centre, Darwin.

Franklin DC (1999) Evidence of disarray amongst granivorous bird assemblages in the savannas of northern Australia, a region of sparse human settlement. *Biological Conservation* **90**: 53-68.

Myers B, Allan G, Bradstock R, Dias L, Duff G, Jacklyn P, Landsberg J, Morrison J, Russell-Smith J and Williams RJ **(**2004) *Fire management in the rangelands*. Report to the Australian Government's Department of Environment and Heritage, Canberra. Tropical Savannas Management Cooperative Research Centre: Darwin. (http://www.deh.gov.au/land/publications/rangelands-fire)

Rossiter N, Setterfield S, Douglas M, Hutley L and Cook G (2004) Exotic grass invasion in the tropical savannas of northern Australia: ecosystem consequences. In: *Proceedings of the 14th Australian weeds conference*. pp. 168–171. Weeds Society of New South Wales, Wahroonga, NSW.

Russell-Smith, J and Bowman DMJS (1992) Conservation of monsoon rainforest isolates in the Northern Territory, Australia. *Biological Conservation* **59**, 51-63.

Russell-Smith J, Ryan PG and Cheal D (2001) Fire regimes and the conservation of sandstone heath in monsoonal northern Australia: frequency, interval, patchiness. *Biological Conservation* **104**, 91-106.

Russell-Smith J, Ryan PG, Klessa D, Waight G and Harwood R (1998) Fire regimes, fire-sensitive vegetation, and fire management of the sandstone Arnhem Plateau, monsoonal northern Australia. *Journal of Applied Ecology* **35**, 829-846.

Russell-Smith J and Stanton JP. 2002. Fire regimes and fire management of rainforest communities across northern Australia. In: *Flammable Australia* (Eds RA Bradstock, J Williams & AM Gill), pp. 329-350. Cambridge University Press: Cambridge,UK.

Russell-Smith J, Yates CP, Edwards A, Allan GE, Cook GD, Cooke P, Craig R, Heath B and Smith R (2003) Contemporary fire regimes of northern Australia: change since Aboriginal occupancy, challenges for sustainable management. *International Journal of Wildland Fire* **12**, 283-297.

Whitehead PJ and McGuffog T (1997) Fire and vegetation pattern in a tropical floodplain grassland: a description from the Mary River and its implications for wetland management. In: *Bushfires '97 Prooceedings*. CSIRO: Darwin.

Williams RJ, Griffiths AD and Allan GE (2002) Fire regimes and biodiversity in the wet-dry tropical landscapes of northern Australia. In: *Flammable Australia: the fire regimes and biodiversity of a continent* (Eds RA Bradstock, JE Williams and AM Gill), pp. 281-304. Cambridge University Press: Cambridge, UK.

Woinarski JCZ, Milne D and Wanganeen G (2001). Changes in mammal populations in relatively intact landscapes of Kakadu National Park, Northern Territory, Australia. *Austral Ecology* **26**, 360-370.

Appendix C. Weeds as a threatening process in the NT

1. Threat Scores

Introduction

Naturalised foreign plants are recognised as major threats to biodiversity (and other values) across the Northern Territory (Smith 2002), and throughout the world. Their impacts can be diverse: at one extreme they may transform environments (for example, over very extensive areas, *Mimosa pigra* may change floodplain grasslands to impenetrable monospecific shrubland thickets), but they may also have impacts that are less extreme or less conspicuous, for example including alteration of fire regimes, reduction in seed and nectar resources for native animals, altering hydrology and soil properties, poisoning stock and native animals, and out-competing native plants (Fairfax & Fensham 2000).

In a recent review, Martin *et al*. (2006) listed 160 exotic plant species considered to be a current threat to Australia's rangeland biodiversity. Each weed species is individual and each will have different impacts, extend over different areas, and be differentially capable of control. Hence it is not possible to readily compile a composite assessment of biodiversity impact of weeds in general. Rather, here we assemble information and assessments for 20 different weed species. These were selected as those that rated most highly (for risk to biodiversity) in the Northern Territory's current weed risk assessment process (as at September 2007: see http://www.nt.gov.au/nreta/natres/weeds/risk). It is recognised that other weed species may have at least local impacts on biodiversity in the Northern Territory, and that the Territory may be exposed in the future to additional significant weed species. The species included are listed in Table C1. Note that many of these species are recognised as serious threats to biodiversity elsewhere in Australia (and indeed elsewhere in the world). For example, Grice (2006a) listed 15 principal weeds threatening biodiversity in the rangelands of Queensland: 8 species are common to these two lists.

Note that this list includes many species not currently declared or proscribed as weeds in the Northern Territory. "Weediness" is a status that may vary depending upon perspectives. At least some species that have significant detrimental impacts upon biodiversity and other environmental values are reported to have beneficial impacts for some industries, most particularly pastoralism (Christian 1959, Mott 1986). These competing valuations have led to some appreciable conflicts in the regulation and management of at least some exotic plant species in the Northern Territory (Lonsdale 1994, Low 1997, Whitehead 1999, Whitehead & Dawson 2000, Whitehead & Wilson 2000, Paynter *et al*. 2003, Grice *et al*. 2006, Cook & Dias 2006, Friedel *et al*. 2006).

Weeds occur throughout the Territory and on lands of all tenures, notably including lands whose primary function is biodiversity conservation (e.g. Cowie & Werner 1993). Generally the extent and diversity of weeds is lowest in most remote, isolated and unmodified areas (e.g. Fensham & Cowie 1998). There are now more than 200 naturalised exotic plants in the Top End of the Territory, making up about 7% of its total flora (Woinarski *et al*. 2007a). While this is a very substantial exotic element for such a relatively natural landscape, it is appreciably less than that for Australia as a whole (2700 naturalised exotic plants, or about 12% of the Australian flora).

Weeds are recognised as a major threat to the Territory's biodiversity. Of 203 Territory species listed as threatened under Australian or Territory legislation, for about 70 species weeds are considered an explicit threat (Woinarski *et al*. 2007b). The only factor considered to affect more Territory threatened species is inappropriate fire regime (see Section 4 of main report).

Table C1. List of foreign plant species considered in this assessment. Weed status: WONS= one of the 20 recognised weeds of national significance; 100 World's Worst=included in the list of 100 of the world's worst invasive alien species (Lowes *et al*. 2000); NT status: class A/C=to be eradicated; class B&C=growth and spread to be controlled; class C=introduction of species is prohibited.

Species	**Common name**	**Life form**	**Weed status**
Acacia nilotica	prickly acacia	shrub	WONS; Class A/C
Andropogon gayanus	gamba grass	grass	Class A/B
Azadirachta indica	neem tree	tree	
Cabomba caroliniana	Cabomba	aquatic herb	Class A/C
Cenchrus ciliaris	buffel grass	grass	
Cryptostegia grandiflora	rubber vine	vine	WONS; Class A/C
Hymenachne amplexicaulis	olive hymenachne	grass	WONS; Class B/C
Jatropha gossypiifolia	bellyache bush	shrub	Class B/C
Lantana camara	Lantana	shrub	WONS; Class B/C; 100 World's Worst.
Leucaena leucocephala	coffee bush	tree	100 World's Worst
Megathyrsus maximus	guinea grass	grass	
Mimosa pigra	Mimosa	shrub	WONS; 100 World's Worst; Class B/C
Parkinsonia aculeata	Parkinsonia	shrub	WONS; Class B/C
Pennisetum polystachion (and *P. pennisetum*)	mission grasses	grass	Class B/C
Prosopis spp.	Mesquite	shrub	WONS; Class A/C; 100 World's Worst
Salvinia molesta	Salvinia	aquatic herb	WONS; Class B/C
Schinus terebinthifolius	Brazilian pepper-tree	tree	100 World's Worst
Tamarix aphylla	athel pine	tree	WONS; Class B/C
Themeda quadrivalvis	grader grass	grass	Class B/C
Urochloa mutica	para grass	grass	

As with some other threats considered here, the impacts upon biodiversity of weeds may vary over time and depending upon the extent and appropriateness of management actions. For example, rubber vine is a major environmental weed in most riparian areas in the Queensland Gulf country, but is currently moving only slowly westwards such that it is not currently a major threatening factor in Territory riparian systems. However, it has a real potential to become a major threat. Where possible, and consistent with the environmental weed risks determined through the Northern Territory weed risk assessment process, we have scored both the current and potential impacts for all 20 weed species considered.

Weeds do not act alone in their impacts upon biodiversity. Rather, they may have complex and compounded inter-relationships with other threatening processes, most particularly including fire, land clearing, pastoralism, feral animals and climate change. Particularly for some invasive pasture species (such as gamba grass, mission grass and buffel grass), the environmental impacts operate mainly through the grass biomass forcing a substantial shift to a more intense and destructive fire regime (e.g. Butler & Fairfax 2003, Rossiter *et al*. 2003). Many of the detrimental impacts upon biodiversity of pastoralism may be compounded if this also involves deliberate replacement of native grasses by exotic ones. Much land clearing is to make habitat more suitable for these exotic grasses.

Assessment of Impact

The Northern Territory Weed Risk Assessment process has provided a collation of all available information on a wide range of exotic plant species, including all species considered here. This information base (not yet publicly accessible) was used as the basis for scoring weed impacts, distribution and habitat. Note that compared to some other threatening factors, for many of the weed species considered here there has been no or relatively little research that has sought to document their impacts upon biodiversity. As such, the scores given here generally represent interpretations from the best available information rather than explicit quantitative evaluations. There are some systematic frameworks available for the assessment of the environmental impacts of weeds (e.g. Adair & Groves 1998), however for most species considered here, there is insufficient information to adequately parameterise such models.

In addition to impacts upon biodiversity, we consider here impacts of weeds upon other values including landscape function, production and cultural values. Weeds may alter landscape function in many ways, for example through enriching or draining soil fertility (Peake *et al*. 1990, Schmidt & Lamble 2002), altering hydrological functioning (particularly so for some aquatic weeds: e.g. Clarkson 1995, 2001, Douglas *et al*. 1998, Douglas & O'Connor 2003) or influencing fire regimes. Exotic plants may affect agricultural production (including also in the case of some aquatic weeds, fisheries production) in diverse ways. Some environmental weeds are championed as bringing great benefit to pastoralism enterprises, although in some cases these same species may over the long-term deprive the soils of their fertility (Kaur *et al*. 2005, Friedel *et al*. 2006). In general, here we have given a score of 0 for impacts on production to those introduced plants that are held to be beneficial to agriculture. The impacts of weeds on cultural values have not been well documented to date, but relate mostly to reductions in the availability of bush tucker or the efficiency with which it can be harvested, to aesthetics and to physical harm (e.g. through weeds with poisons or spines): most scoring here for this variable is interpreted from Smith (2001) and Gardener (2005).

2. The costs of weed management

Assessment of Management costs

There is no simple look-up table already available for providing the costs required to control or eliminate weeds in the Northern Territory, although some indicative figures are available for some weed species. Costs will vary according to the conspicuousness of the weed species, the remoteness and accessibility of a location, the type of control and the susceptibility of the weed species to that control mechanism, the incidence of the weed species in the broader region generally, and to the dispersal, ecological, life history and demographic characteristics of the weed species. In general, it will be most cost-effective if weeds can be prevented from invading in the first place, then if initial incursions are eliminated quickly, then by dealing with satellite populations. In common with the management of feral animals, half-hearted attempts that do little other than temporarily reducing local infestations may be effectively useless and simply waste money, resources and goodwill.

For environmental weeds that may also bring economic benefit to some stakeholders (such as buffel grass), there is a reasonable argument that those benefiting from the spread of those exotic plants on their property should particularly contribute to the costs of containment, should the plant escape to neighbouring lands where it is unwanted: the "polluter pays" principle (Grice 2006b); although this has yet to be implemented. Otherwise, across the Territory and depending upon the gazetted status of a particular weed, the cost of weed management is largely the responsibility of the landholder, albeit typically with access to management funding from a range of Territory and Australian government sources.

For some weed species, or for particular weeds in some areas, eradication may be infeasible. For example, Kean & Price (2003) considered that for mission grasses *Pennisetum* spp. in the Darwin peri-urban area,

> "the prospects of significantly reducing (its) prevalence … are very limited indeed. Probably the only success that can be achieved is to reduce it on a small scale. Control actions may need to be repeated regularly and indefinitely because seed sources outside the target area might never be fully eliminated."

For this report, we attempt to base costings of weed management on the contextual target provided in the *Integrated Natural Resource Management Plan for the Northern Territory*,

> "RCT3-4 By 2020, there is strategic containment of declared weeds, ecologically invasive plants and feral animals, sufficient to ensure that they have no significant impact on the conservation status of any Territory species or ecological community."

Where possible, we base our costings here on estimates derived in the feasibility of control modules of the Territory weed risk assessment process. This process includes two costings, both based on the estimated cost to treat a one hectare infestation of the weed in the first year of targeted control (for an infestation that has reached maximum weed density) (Table C2). One cost is for the price of the amount of chemical necessary to control that 1 ha of infestation (scores as either very high (>$500/ha), high ($250-$500/ha), medium ($100-$250/ha) or low (<$100/ha). The other available figure is an estimate of labour costs, categorised on the same four class scale.

Note that these figures do not include costs of locating the weed or travelling to the site. Further, in some cases, management or eradication may be better achieved by biological control and/or fire management and/or pastoralism management rather than chemicals alone (or better – a combination of a range of integrated control mechanisms: Vitelli & Pitt 2006). Other weed control costs are associated with training (e.g. in weed recognition and correct use of chemicals), surveillance and administration (e.g. for the organisation of Indigenous ranger groups). In many cases, the costs of these necessary management components, and of alternatives to chemical control, are not considered here, because there is too little information available about their development costs, success and duration. One indication of training costs is given in Sinden *et al*. (2003) where training for Indigenous rangers and others for mimosa control and management was costed at $3.27 million over a 4.8 year period (September 1998 to June 2003).

Table C2. Costs of control (for a located 1 ha infestation at maximum density) for all weed species considered in this assessment. Based on figures supplied from the NT weed risk assessment process. Codes for chemical and labour costs: VH (>$500/ha), H ($250-$500/ha), M ($100-$250/ha), L (<$100/ha).

Species	Common name	Cost of chemical treatment	Labour costs
Acacia nilotica	prickly acacia	H	M
Andropogon gayanus	gamba grass	M	H
Azadirachta indica	neem tree	M	M
Cabomba caroliniana	Cabomba	H	H
Cenchrus ciliaris	buffel grass	L	M
Cryptostegia grandiflora	rubber vine	VH	VH
Hymenachne amplexicaulis	olive hymenachne	M	VH
Jatropha gossypiifolia	bellyache bush	M	H
Lantana camara	Lantana	M	M
Leucaena leucocephala	coffee bush	VH	H
Megathyrsus maximus	guinea grass	M	H
Mimosa pigra	Mimosa	M	M
Parkinsonia aculeate	Parkinsonia	H	M
Pennisetum polystachion (and *P. pennisetum*)	mission grasses	M	H
Prosopis spp.	Mesquite	M	H
Salvinia molesta	Salvinia	M	M
Schinus terebinthifolius	Brazilian pepper-tree	VH	VH
Tamarix aphylla	athel pine	VH	VH
Themeda quadrivalvis	grader grass	M	M
Urochloa mutica	para grass	M	VH

Some recent reviews have attempted to collate total costs of weed management projects and related these to the resulting containment or eradication outcomes (Sinden *et al*. 2003, Gardener *et al*. 2004, Martin & van Klinken 2006). These studies provide far more detail for some of the species considered here; and also provide some case studies of the cost-efficiency of a range of weed management options for some species. Sinden *et al*. (2003) attempted to model appropriate expenditure on weed control related to the environmental damage (in this case, the number of threatened species affected) due to the weed species. Their econometric model suggested that an appropriate increase in expenditure for the management of a weed species was $68,700 per year for each threatened native species affected by that weed.

Martin & van Klinken (2006) collated estimates of total expenditure by the Australian government (through NHT and the National Action Plan for Salinity) aimed at management of individual weeds of national significance over an 8-year period (Table C3). Martin & van Klinken (2006) estimated that the Northern Territory received about $7.5 million over the 5-year period 2000-05 from these sources for weed management. Territory government funding for weed management includes about $4.7 million/year of employment and operational expenses for rangers on Territory parks (excluding Kakadu and Uluru) and $2.3 million for staff and operational expenditure for the weeds branch of NRETA (Martin & van Klinken 2006). Kakadu spends about $0.5 mlllion/year on mimosa control, and a further $0.1 million/year on control of invasive exotic grasses (Martin & van Klinken 2006). In addition, many pastoral enterprises and mining companies may spend considerable amounts on weed management on lands under their control: for example, Tipperary pastoral station spends about $1.5 million/year on control of mimosa alone (B. Rankmore, *pers. comm*.).

Table C3. Total expenditure by the Australian government aimed at on-ground management of some rangeland weeds of national significance.

Species	**Total ($ million, over the period 1997-2005)**
Athel pine	$0.82
cabomba	$0.61
olive hymenachne	$1.19
Lantana	$2.45
mesquite	$2.02
Mimosa	$5.77
Parkinsonia	$2.03
Prickly acacia	$1.75
Rubber vine	$1.68
Salvinia	$0.96
TOTAL	**$19.29**

In a case study of mimosa control, Gardener *et al*. (2004) estimated that a total of at least $50 million has been spent (mostly by the Territory and Australian governments) since the early 1980s. For several regions on Aboriginal lands, Gardener *et al*. (2004) notes success in reducing the extent of mimosa infestations (from 8800 to 3300 ha in the regions considered) over a 5 year period (with $10 million expenditure). Factors leading to successful reduction were initial (then annual) broad-scale helicopter-based spraying, followed by on-ground spraying targeting emergent seedlings and stands missed by aerial spraying; and strategic approached concentrating on upstream and peripheral populations. An important part of the success of the program was surveillance and the elimination of new infestations in areas distant from the main infestations, notably in remote central and south-east Arnhem Land.

3. References

Adair RJ and Groves RH (1998) *Impact of environmental weeds on biodiversity: a review and development of a methodology*. National Weeds Program, Environment Australia, Canberra.

Butler DW and Fairfax RJ (2003) Buffel Grass and fire in a Gidgee and Brigalow woodland: a case study from central Queensland. *Ecological Management & Restoration* **4**, 120-125.

Christian CS (1959) The future revolution in agriculture in Northern Australia. *Australian Journal of Science* **22**, 138-147.

Clarkson J (1995) Ponded pastures: a threat to wetland biodiversity. In *Wetland research in the wet-dry tropics of Australia*. Supervising Scientist Report 101.

Clarkson J (2001) From controversy to a new awareness: experience from two decades of ponded pasture in Queensland. In *Wise use of wetlands in northern Australia: grazing management in wetlands and riparian habitats*. (eds B. Myers, M. McKaige, P. Whitehead, and M. Douglas) p. 33. Centre for Tropical Wetlands Management, Northern Territory University: Darwin.

Cook GD and Dias L (2006) It was no accident: deliberate plant introductions by Australian government agencies during the 20th century. *Australian Journal of Botany* **54**, 601-625.

Cowie ID and Werner PA (1993) Alien plant species invasive in Kakadu National Park, tropical northern Australia. *Biological Conservation* **63**, 127-135.

Douglas MM and O'Connor RA (2003) Effects of the exotic macrophyte, para grass (*Urochloa mutica*), on benthic and epiphytic macroinvertebrates of a tropical floodplain. *Freshwater Biology* **48**, 962-971.

Douglas M, Finlayson CM and Storrs, MJ (1998) Weed management in tropical wetlands of the Northern Territory, Australia. In *Wetlands in a dry land: understanding for management* (ed. W.D. Williams.) pp. 239-252. Environment Australia, Canberra.

Fairfax RJ and Fensham RJ (2000) The effect of exotic pasture development on floristic diversity in central Queensland, Australia. *Biological Conservation* **94**, 11-21.

Fensham RJ and Cowie ID (1998) Alien plant invasions on the Tiwi Islands: extent, implications and priorities for control. *Biological Conservation* **83**, 55-68.

Friedel M, Puckey H, O'Malley C, Smyth A and Miller G (2006). *Buffel grass: both friend and foe. An evaluation of the advantages and disadvantages of buffel grass use and recommendations for future research*. Report 17. Desert Knowledge Cooperative Research Centre, Alice Springs.

Gardener M (2005). *Towards more strategic management of weeds on Top End Aboriginal lands.* Tropical Savannas Cooperatoive Research Centre, Darwin, and James Cook University, Townsville.

Gardener MR, Storrs MJ and Wingrave S (2004). Towards more strategic management of weeds on Top End Aboriginal lands. *Proceedings of the Fourteenth Australian Weeds Conference*. (eds B.M. Sindel and S.B. Johnson.) pp. 199-202. Weed Science Society NSW, Sydney.

Grice AC (2006a) The impacts of invasive plant species on the biodiversity of Australian rangelands. *Rangelands Journal* **28**, 27-35.

Grice AC (2006b) Commercially valuable weeds: can we eat our cake without choking on it? *Ecological Management and Restoration* **7**, 40-44.

Kaur K, Jalota RK, Midmore DJ and Rolfe J (2005) Pasture production in cleared and uncleared grazing systems of central Queensland, Australia. *Rangeland Journal* **27**, 143-149.

Kean L and Price O (2003) The extent of mission grasses and gamba grass in the Darwin region of Australia's Northern Territory. *Pacific Conservation Biology* **8**, 281-290.

Lonsdale WM (1994) Inviting trouble: introduced pasture weeds in northern Australia. *Australian Journal of Ecology* **19**, 345-354.

Low T (1997) Tropical pasture plants as weeds. *Tropical Grasslands* **31**, 337-343.

Lowe S, Browne M, Boudjelas S and De Poorter M (2000) *100 of the World's worst invasive alien species: a selection from the Global Invasive Species Database* (Invasive Species Specialist Group, World Conservation Union (IUCN), Auckland.)

Martin TG and van Klinken RD (2006) Value for money? Investments in weed management in Australian rangelands. *Rangeland Journal*. **28**, 63-75.

Martin TG, Campbell S and Grounds S (2006) Weeds of Australian rangelands. *Rangeland Journal*. **28**, 3-26.

Mott JJ (1986) Planned invasions of Australian tropical savannas. In *Ecology of biological invasions: an Australian perspective* (eds R.H. Groves and J.J. Burdon.) pp. 89-96. Australian Academy of Science, Canberra.

Paynter Q, Csurhes SM, Heard TA, Ireson J, Julien MH, Lloyd J, Lonsdale WM, Palmer WA, Sheppard AW and van Klinken RD (2003) Worth the risk? Introduction of legumes can cause more harm than good: an Australian perspective. *Australian Systematic Botany* **16**, 81-88.

Peake DCI, Myers RJK and Henzell EF (1990) Sown pasture production in relation to nitrogen fertilizer and rainfall in southern Queensland. *Tropical Grassland* **24**, 291-298.

Rossiter NA., Setterfield SA, Douglas MM and Hutley LB (2003) Testing the grass-fire cycle: alien grass invasion in the tropical savannas of northern Australia. *Diversity and Distributions* **9**, 169-176.

Rossiter N, Setterfeld S, Douglas M, Hutley L and Cook G (2004) Exotic grass invasion in the tropical savanna of northern Australia: ecosystem consequences. *Proceedings of the Fourteenth Australian Weeds Conference*. (eds B.M. Sindel and S.B. Johnson.) pp. 168-171. Weed Science Society NSW, Sydney.

Schmidt S and Lamble RE (2002) Nutrient dynamics in Queensland savannas: implications for the sustainability of land clearing for pasture production. *Rangeland Journal* **24**, 96-111.

Sinden J, Jones R, Hester S, Odom D, Kalisch C, James R and Cacho O (2003) *The economic impact of weeds in Australia*. Technical series 8. CRC for Australian weed management, Adelaide.

Smith N (2001) *Not from here: plant invasions on Aboriginal lands*. Tropical Savannas Cooperative Research Centre: Darwin.

Smith NM (2002) *Weeds of the wet/dry tropics of Australia: a field guide*. Environment Centre, Darwin.

Vitelli JS and Pitt JL (2006) Assessment of current weed control methods relevant to the management of biodiversity of Australian rangelands. *Rangeland Journal* **28**, 37-46.

Whitehead P (1999) Promoting conservation in landscapes subject to change: lessons from the Mary River. *Australian Biologist* **12**, 50-62.

Whitehead P and Dawson T (2000) Let them eat grass! *Nature Australia* **Autumn 2000**, 46-55.

Whitehead P and Wilson C (2000) Exotic grasses in northern Australia: species that should be sent home. *Proceedings of the Northern Grassy Landscapes Conference, Katherine, Northern Territory*. pp. 1-7. Tropical Savannas CRC, Darwin.

Woinarski J, Mackey B, Nix H and Traill B (2007a) *The Nature of Northern Australia*: *natural values, ecological processes and future prospects*. (ANU e-press, Canberra.)

Woinarski J, Pavey C, Kerrigan R, Cowie I and Ward S (2007b) *Lost from our landscape: threatened species of the Northern Territory*. NT Government Printer, Darwin.

Appendix D: Feral Animals as a Threatening Process in the NT

1. A review of threats

Habitat degradation, competition and other impacts of introduced herbivores

European rabbit (*Oryctolagus cuniculus*)

European rabbits occur over the southern two thirds of the Northern Territory. To the north of Alice Springs, rabbit distribution is extremely patchy with rabbits restricted mainly to calcareous soil or limestone outcrops (Low & Strong 1983). South of Alice Springs, rabbit distribution is also patchy but rabbits occupy a boader range of land system types (Low & Strong 1983). Rabbits have had a profound effect on the vegetation of the rangelands (Lange & Graham 1983, Foran *et al*. 1985, Williams *et al*. 1995). Perennial plant species have been replaced by annuals largely as a result of grazing by rabbits in many areas (Hall *et al*. 1964), and until recently, the recruitment of palatable shrubs and trees was suppressed by rabbits over vast expanses of the arid rangelands (Lange & Graham 1983, Foran *et al*. 1985). Several authors (Morton 1990, Williams *et al*. 1995, Woinarski 2001) suggest that rabbits have played a key role in the demise of arid zone mammals whether indirectly by supporting high populations of introduced predators (e.g. feral cats *Felis catus*, foxes *Vulpes vulpes*) or directly through competition and habitat degradation. Robley *et al*. (2002) suggest that the former of these mechanisms has had by far the greater impact on native mammals. Competition with native fauna and land degradation by European rabbits are listed as key threatening processes under the Environment Protection and Biodiversity Conservation (EPBC) Act (1999). Rabbits also compete with stock for pasture and the economic cost to production industries is estimated to be $113.1M annually (McLeod 2004). However it must be stated that most of this cost is incurred in areas other than the Northern Territory where rabbits still occur at high densities even with RHD.

Feral horse (*Equus caballus*) and feral donkey (*E. asinus*)

Recent aerial surveys suggest that there are about 265,000 feral horses and 165,000 feral donkeys in the Northern Territory (K. Saalfeld, Parks and Wildlife Service of the Northern Territory, unpublished data, 1986-2001). Feral horses and feral donkeys are patchily distributed within the Northern Territory (Dobbie *et al*. 1993, Wilson *et al*. 1992). There are major concentrations in the Victoria River District, Arnhem Land, the Gulf and the Darwin region. Horses and donkeys also occur in the Central region (between Tennant Creek and Alice Springs) and around Alice Springs. Although the environmental impacts of feral horses are not well documented, it is believed that they contribute to erosion, damage vegetation and disperse weeds (Dobbie *et al*. 1993).

Unequivocal data on the competitive impacts of feral horses on native animals are lacking. The economic cost to production industries for feral horses alone is estimated to be $0.5M annually (McLeod 2004).

Water buffalo (Bubalus bubalis)

Feral water buffalo are confined to the northern third of the Northern Territory. Prior to the national Brucellosis and Tuberculosis Eradication Campaign (BTEC), which saw widespread elimination of populations, there were approximately 340,000 water buffalo in northern Australia (Bayliss & Yeomans 1989a,b). The current population of feral water buffalo in the Northern Territory has been estimated at about 73,000 (K. Saalfeld, Parks and Wildlife Service of the Northern Territory, unpublished data, 1997-2000). In the absence of management designed to regulate numbers, the population of feral water buffalo is likely to return to pre-BTEC levels.

Disturbance by feral water buffalo may be most concentrated in remnant monsoon forest patches where sapling damage results in the suppression of recruitment in some tree species (Braithwaite *et al*. 1984), and in coastal floodplain wetlands. In the latter environments, buffalo may trigger major changes because their disturbance may facilitate saltwater intrusion (Braithwaite *et al*. 1984, Whitehead *et al*. 1990). Where present in high densities in eucalypt forests more generally, buffalo may have very substantial impacts on ecosystem processes: for example, Werner *et al*. (2006) noted that buffalo “initiate a cascade of effects by changing ground-level biomass, which change competitive relationships and fuel loads, which then have an impact on tree growth and demography”.

Feral camel (Camelus dromedarius)

Feral camels are widely distributed in the rangelands of Western Australia, South Australia and the Northern Territory (Short *et al*. 1988). In the Northern Territory, feral camels are mainly confined to the southern third of the land area. A 2001 aerial survey indicted that there was a minimum of 80,500 feral camels in the Northern Territory (Edwards *et al*. 2004) and that the population is doubling every eight years (Edwards *et al*. 2004). Feral camels are believed to contribute to soil erosion, damage vegetation and foul waterholes (Dörges & Heucke 2003, P. Latz, pers. comm.). Feral camels also damage infrastructure on pastoral properties and compete with stock for pasture. Feral camels also have demonstrable impacts on Aborigional cultural values, including water holes and artefact and bushtucker resources.

Feral pig (Sus scrofa)

In the Northern Territory, feral pigs are mainly confined to the northern third of the landmass, including Bathurst Island. Reliable population estimates are difficult to obtain for feral pigs; but in the mid-1990s there were estimated to be 3.5 - 23.5 million feral pigs in Australia (Choquenot *et al*. 1996). Feral pigs are omnivorous and consume a wide range of plants and animals (Choquenot *et al*. 1996). Feral pigs also root up ground contributing to soil erosion and river bank destabilisation, and can locally threaten some plant species (e.g. geophytes with edible roots, Russell-Smith & Bowman 1992, Choquenot *et al*. 1996). The economic cost to production industries for feral pigs is estimated to be $106.5M annually (McLeod 2004). Feral pigs also have the potential to act as a vector for numerous livestock and wildlife diseases (Choquenot *et al*. 1996). Accordingly, predation, habitat degradation, competition with native fauna and disease transmission by feral pigs is listed as a key threatening process under the EPBC Act.

Predation

Feral cat (Felis catus)

Feral cats are distributed throughout the Northern Territory. Although densities as high as 6.3 km^{-2} have been recorded in the Mitchell grass downs east of Tennant Creek during an eruption of the long-haired rat (*Rattus villosissimus*) (G. Edwards, Parks and Wildlife Service of the Northern Territory, unpublished data, 1994), densities in the order of 0.1 - 0.6 km^{-2} are more typical, at least in the southern rangelands (Jones & Coman 1982, Edwards *et al*. 2001). Predation by feral cats is listed as a key threatening process under the EPBC Act. The clearest evidence that cat predation can have a serious impact on native fauna comes from recent attempts to reconstruct rangeland mammal assemblages (Dickman 1996). Predation by feral cats has hampered attempts to reintroduce the rufous hare-wallaby (*Lagorchestes hirsutus*) in central Australia (Gibson *et al.* 1994), the burrowing bettong (*Bettongia lesueur*) and numbat (*Myrmecobius fasciatus*) in Western Australia (Christensen & Burrows 1995, Friend & Thomas 1995) and the brush-tailed bettong (*Bettongia penicillata*) in New South Wales (D. Priddel, New South Wales National Parks and Wildlife Service, unpublished data, 2002).

Red fox (Vulpes vulpes)

Foxes are distributed across the southern half of the Northern Territory. Reported densities in similar habitats in other states are in the range 0.6 - 2 km^{-2} (Marlow 1992, Saunders *et al*. 1995) with densities being higher in areas with rabbits but without dingoes (Saunders *et al*. 1995). Foxes were rarely encountered in the Tanami Desert northwest of Alice Springs in the 1970s and early 1980s (Bolton & Latz 1978, Gibson 1986), but now they are relatively common there as far north as Tennant Creek (Paltridge & Southgate 2001). There is abundant evidence that predation by foxes is a major threat to native fauna (Burbidge & McKenzie 1989). Foxes have been shown to have a major detrimental impact on existing populations of black-footed rock-wallabies (*Petrogale lateralis*: Kinnear *et al*. 1988, 1998), brush-tailed bettongs (Saunders *et al*. 1995), numbats (Friend 1990) and tammar wallabies (*Macropus eugenii*: Saunders *et al*. 1995) in Western Australia. Populations of all these animals increased following intensive fox control. One of the two last known wild populations of the rufous hare-wallaby in the Northern Territory was exterminated by a single fox (Lundie-Jenkins *et al*. 1993). Predation by foxes is appropriately listed as a key threatening process under the EPBC Act.

<u>Wild dog (Canis lupus dingo and Canis lupus familiaris)</u>

The term “wild dog” applies to two sub-species of canid; the dingo (*Canis lupus dingo*) and the feral domestic dog (*C. l. familiaris*) and hybrids of the two.

The dingo has inhabited Australia for about 4000 years, long enough to become a functional part of the natural ecological system as a top order predator. In view of their ecological importance and totemic status with some Aboriginal groups, dingoes are regarded under Northern Territory (NT) legislation as native wildlife. This status affords the dingo full legal protection, making it an offence to possess, interfere with, or kill dingoes unless authorised to do so under the Territory Parks and Wildlife Conservation Act (2000) (TPWCA).

Domestic dogs were introduced to the NT with European settlement and populations of feral domestic dogs and dingo/domestic dog hybrids are known to exist in the vicinity of human habitation. Outside these areas, the wild dog population is thought to comprise largely pure dingoes.

There are a number of negative or undesirable impacts associated with dingoes and other wild dogs:

1.) They are known predators of livestock and they can cause significant economic losses to pastoral production.
2.) They can be a menace to tourists and staff at remote tourist resorts and national parks.
3.) They can have an impact on the survival of remnant populations of endangered fauna, and
4.) In high rainfall areas (mainly east of the Great Divide), they are implicated in the spread of parasites (hydatids and neospora) that affect cattle and cause significant losses to beef production.

The Northern Territory Cattlemen’s Association (NTCA) recently put the cost of dingo depredations to the pastoral industry in the NT at $2 million annually.

Although there are few benefits associated with feral domestic dogs and hybrids, there are several advantages in maintaining wild populations of pure dingoes in the NT.

Dingoes are an important part of the natural ecological system in Australia. They are thought to regulate populations of some native species that could otherwise be pests, such as kangaroos and wallabies. Kangaroo populations in parts of Australia where dingoes have been eradicated are up to 5 times higher than in areas where dingo populations are intact. Dingoes also prey upon introduced pest species such as rabbits, pigs, foxes and feral cats which may help to keep their numbers in check. The fact that there are no feral goats in the NT is directly attributable to the presence of the dingo. Feral goats are estimated to cost the pastoral industry across Australia a net amount of $17.8 million annually through reduced stock production. Production losses attributable to kangaroos are probably just as high in areas where dingoes have been eradicated.

Cross-breeding with domestic dogs represents a significant threat to the long-term persistence of pure dingoes in Australia. The level of hybridisation in dingo populations in south-eastern Australia is very substantial; in contrast, the genetic integrity of dingoes in the NT remains largely intact, affording them significant conservation value.

Habitat degradation, competition and poisoning due to cane toads

The cane toad (*Bufo marinus*) was introduced to Queensland in 1935 and is still spreading to the west and south. Cane toads now occur across most of the top third of the mainland Northern Territory. Cane toads have also colonised some of the islands of the Northern Territory, notably those in the Sir Edward Pellew group. The impacts of cane toads are environmental, economic and cultural. The cane toad is poisonous in all its life stages. Populations of varanid lizards and snakes show marked declines when cane toads first colonise an area, primarily as a result of poisoning (McRae *et al*. 2005). These are important bushtucker species for Aboriginal people and the basis of economic enterprise in some areas (McRae *et al*. 2005). However, populations rarely decline to extinction and generally recover to some extent. The most compelling evidence for catastrophic decline is for the northern quoll which has been shown to decline to local extinction in some areas when cane toads arrive (Oakwood 2004, Watson and Woinarski 2003). While it is suspected that habitat degradation and competition due to cane toads may have major impacts on invertebrates, other frogs and some reptiles, no compelling evidence exists (McRae *et al*. 2005). The cane toad is listed as a key threatening process under the EPBC Act.

Assessment of impact

Vegetation condition

Rabbits, camels, horses, donkeys, pigs and buffalo were considered to have a direct impact on vegetation condition through grazing and trampling in areas where they occurred. The level of impact was assumed to increase with density of the feral pest. Foxes, cats, wild dogs and cane toads were not considered to have an impact on vegetation condition.

Threatened and susceptible species
Rabbits, camels, horses, donkeys, pigs and buffalo were considered to have a direct impact on threatened and susceptible plants and plant communities through grazing and trampling in areas where they occurred. The level of impact was assumed to increase with density of the feral pest. These feral species were also considered to have a direct impact on threatened and susceptible animals through habitat modification and competition for food and other resources where they occurred at moderate to high densities. Foxes, cats and wild dogs were considered to have a direct impact through predation on sensitive and threatened animals throughout their ranges. Cane toads were considered to have a direct impact through toxic ingestion on many predators throughout their range. Cane toads were not considered to have an impact on their prey or on other frog species.

Landscape function

Rabbits, camels, horses, donkeys, pigs and buffalo were considered to have a direct impact on landscape function through grazing and trampling in areas where they occurred. The level of impact was assumed to increase with density of the feral pest. Foxes, cats, wild dogs and cane toads were not considered to have an impact on landscape function.

Production

Rabbits, camels, horses, donkeys, pigs and buffalo were considered to have a direct impact on production through competiton for food and habitat modification in areas where they overlapped with pastoralism, agriculture and bush food production. The level of impact was assumed to increase with density of the feral pest. Rabbits, camels, horses, donkeys, pigs and buffalo were considered to have an indirect impact (through spread of weeds) on production in areas wherever they overlapped with pastoralism, agriculture and bush food production. Foxes, cats and cane toads were not considered to have an impact on production. Wild dogs were considered to have a direct impact on pastoral production in areas where they overlapped but in the case of the dingo, there are also indirect benefits to production (see above).

Culture

Rabbits, camels, horses, donkeys, pigs and buffalo were considered to have a direct impact on culture across their ranges through habitat modification, damage to culturally important sites, and damage to cultural resources like bush foods and trees used for atrefact production. The level of impact was assumed to increase with density of the feral pest. Foxes, cats and cane toads were considered to have a direct impact on culture wherever they occurred through loss of totemic animal species and reduction in the availability of bush foods (eg, goanna).

2. Economics of feral animal control costs in the NT

Summary

A cost-benefit analysis is carried out with regard to feral animal control activities and the direct potential benefits of such activities. Based on expert opinion obtained through a series of workshops, and with a view to achieving the NT INRM Plan goal by 2020, specific control strategies for all the main feral species in the NT were identified. These species include camels, horses, donkeys, buffalos, pigs, dogs, cats, foxes and cane toads.

Two different aerial control strategies were modelled for large ruminants and pigs, one which involved annual culling and a second which involved periodic culling only when a specific feral animal density was reached. Trapping/baiting, eradication and exclusion strategies were modelled for the remaining species. The direct economic benefits for the pastoral industry of large ruminant control were modelled. Although environmental and cultural values are also likely to be important, their modelling was beyond the scope of this study.

While the control program costs and net benefit estimates derived above are entirely dependent on the data provided by the relevant experts in this field, it is argued that the results obtained provide useful “ball park” figures upon which policy recommendations and the identification of future research priorities can be identified. The robustness of the findings are explored through a number of sensitivity analyses covering such issues as the degree to which feral animals compete with livestock for grazing resources, feral population growth rates, aerial shooting costs, infrastructure constraints, discount rates and time horizons.

The main findings are that the total present costs of a control program for all the species considered would be in excess of $615.5m over a 20 year time horizon (at a 5% discount rate). This is equivalent to an annualised present cost of $49.4m. Rabbits and pigs contribute to 94.4% of this cost, large ruminants 4.6% and the remaining species 1.0% of the total.

However, given the fact that the costs of controlling rabbits and pigs dominate the overall results and the difficulties/uncertainties involved in estimating their control costs, the main focus of the analysis carried out was with regard to large ruminants. The total present costs of the large ruminant control program were approximately $28.1m over a 20 year time horizon (given a 5% discount rate) and equivalent to an annualised present cost of $2.26m.

While such control costs are large, they are far outweighed by the direct economic benefits to the livestock industry from reduced competition between livestock and large feral ruminants. The net present benefits of a control program are thus estimated to be in the region of $180.7m over 20 years, equivalent to an annualised present benefit of $14.5m p.a. Net benefits are likely to be even higher if alternative control

methods (e.g. ground shooting) can be undertaken effectively, if direct economic benefits can be generated from control programs (e.g. use of culled animals for pet meat) and if environmental and cultural values are taken into account. At the same time, a further implication of such findings is that even though environmental and cultural benefits of a control program may be large, their valuation will not affect the overall recommendations resulting from this cost-benefit analysis.

If such significant net benefits can be confirmed in practice (e.g. through a more in-depth study), then there would appear to be a very strong argument for implementing a full-scale feral animal control program in the near future. Furthermore, the magnitude of these direct economic benefits suggests that a control strategy based on annual culling is almost always likely to be preferred.

Finally the analysis indicated that the costs of implementing a control program of the type described in this report, almost always involves large scale expenditure in the first few years (50-75% of total funds over 20 years being spent in the first 5 years). Hence, the annualised figures presented in this report tend to heavily understate the initial funding requirements.

The sensitivity analyses revealed that these “ball park” findings are quite robust in the face of changed assumptions about some of the key variables. Hence, further research to provide a level of detail upon which strong policy recommendations can be confidently made is likely to be highly justified. A more in-depth study is also urgently needed with regard to rabbit and pig control costs.

Project context and aims

The project “Review of threats to biodiversity in the NT” takes place within the context of the 2005 Integrated Natural Resource Management Plan for the Northern Territory (INRM Plan), which states that “by 2020 the extent, condition and functionality of all native Territory environments will be maintained at levels to be set by 2006".

The overall project aim is to inform the implementation of the INRM Plan by prioritising the major threatening processes to biodiversity in the NT. These include bushfires, feral animals, invasive weeds, land clearing and pastoralism.

In addition to taking into account the severity and extent of such threats across NT bioregions, an economic analysis of the magnitude of the costs and benefits of controlling such threats plays an important role in such prioritisation.

This section of the project report focus specifically on the economic analysis of feral animal control in the NT.

Background to feral animal management in the NT

General considerations

a. Eradication is rarely a feasible management objective
b. The best management strategy is to manage in order to reduce impacts and not necessarily population size. However, the two are often highly correlated.
c. Invariably the cost of managing pest animals increases as density decreases.
d. Management should be adaptive. That is, the effort, numbers of ferals and numbers of the affected species should all be monitored, and the results used to fine tune the management program. Monitoring should comprise between 5 and 10% of the total budget.

European rabbit

Australia wide, biological control has had a significant impact on rabbit populations. Myxomatosis, which was deliberately introduced in the early 1950s, had a marked initial impact on rabbit populations. However, populations developed resistance and recovered to varying extents depending on location (Coman 1999). Rabbit Haemorrhagic Disease (RHD) (commonly referred to as rabbit calicivirus disease in Australia and New Zealand), which became established in the wild in Australia in late 1995, has reduced rabbit numbers across much of the rangelands by over 80% (Cooke 1999, Edwards *et al*. 2002a, Neave 1999). The associated decline in the environmental impacts of rabbits has allowed the regeneration of many perennial shrubs and trees (Sandell & Start 1999) that were threatened with extinction across the rangelands (Woinarski 2001). Although there is emerging evidence of resistance in rabbits to RHD in some parts of Australia, rabbit populations in the Northern Territory remain at about 80% of pre-RHD levels. Myxomatosis and RHD are currently endemic in the Northern Territory and the re-release of either virus is not normally considered an option for management. Warren ripping, a non-biological technique which can be applied over large areas, has proved both cost efficient and effective in driving rabbit populations in the Northern Territory to lower levels than would normally occur following outbreaks of myxomatosis and RHD (Edwards *et al*. 2002b). Warren ripping has the added advantage that it destroys the rabbit’s safe harbour thus stemming population increase at times when either myxomatosis or RHD is not active. The implementation of warren ripping in key areas of rabbit habitat (estimated as 10% of the area south of Alice Springs, i.e. 300 x 950 km^2) was considered the ‘best’ option for mitigating the current and potential threat posed by rabbits.

Feral horse and feral donkey

Aerial shooting using a helicopter can be used to humanely and cost effectively control feral horses and donkeys over large areas (Dobbie *et al*. 1993). In some instances, control can be assisted through trapping and/or mustering for the purpose of commercial sale (Dobbie *et al*. 1993). However, the extent to which trapping and mustering can be used depends on market demand and the accessibility of the animals under management. In the Northern Territory, all three techniques are used to manage horses and donkeys. Aerial shooting across highly infested areas was considered the 'best' option for mitigating the current and potential threat posed by horses and donkeys.

Water buffalo

As for horses and donkeys.

Feral camel

As for horses and donkeys.

Feral pig

Trapping, poisoning, aerial shooting using a helicopter and ground shooting are among the methods used to control feral pigs (Saunders 1993; Choquenot *et al*. 1996). Aerial shooting across highly infested areas was considered the 'best' option for mitigating the current and potential threat posed by feral pigs.

Feral cat

Feral cat

Feral cats have been eradicated from islands using a combination of techniques (Veitch 1985; Berruti 1986; van Rensburg *et al*. 1987; van Rensburg & Bester 1988; Bloomer & Bester 1992). However, broadscale control on the Australian mainland has proved problematical (Christensen & Burrows 1995). Currently, the only option potentially available for abating the threat of cat predation over large areas is poison baiting (Short et al. 1997). However, feral cats rarely scavenge (Bayly 1978; Paltridge *et al*. 1997) and it appears that a degree of control can only be achieved by distributing poisoned baits at times of low prey abundance (Short *et al*. 1997). At present, the only registered toxin for feral cats is compound 1080 (sodium monofluoroacetate). This toxin is not cat-specific and there would likely be considerable non-target impacts if it were applied over large areas in an attempt to manage cats. The dingo, which is totally protected in the NT, would be at risk.

Red fox

Foxes can be effectively controlled over large areas using baits containing the compound 1080 (sodium monofluoroacetate) (Christensen & Burrows 1995; Saunders *et al*. 1995; Thomson *et al*. 2000). However, this toxin is not fox-specific and there would likely be considerable non-target impacts if it were applied over large areas in an attempt to manage foxes. The dingo, which is totally protected in the NT, would be at considerable risk.

Wild dog

The Parks and Wildlife Service (PWSNT) is responsible for managing wild dogs outside of town boundaries throughout the NT. The PWSNT dingo and feral dog control program provides a level of protection against economic loss to landholders consistent with the objective of maintaining wild populations of the dingo. Sodium monofluoroacetate (1080) is the approved poison for wild dog control, currently administered in fresh meat baits.

Cane toad

Currently, there is no broadscale control technology that can be successfully applied on mainland Australia to manage cane toads (Macrae et al. 2005). In terms of conserving biodiversity, the best strategy is to prevent cane toads from colonising islands which are suitable for their survival.

Model Description

A cost-benefit analysis is carried out with regard to feral animal control activities and the potential benefits for the pastoralist industry[3]. Based on expert opinion obtained through a series of workshops and with a view to achieving the INRM Plan goal by 2020, specific control strategies for all the main feral species in the NT were identified. These species include: camels, horses, donkeys, buffalos, pigs, dogs, cats, foxes and cane toads. Existing animal population numbers, their dynamics and control cost data were obtained through existing published material and expert opinion.

For each species the discounted stream of future costs and benefits associated, with their respective control strategies over a given time horizon, were calculated. A comparative analysis was then carried out with respect to two different strategies. While both strategies aimed to achieve a large ruminant feral animal density of 0.25 animals/km^2 in the first year and then no more than 0.1/km^2 [4] thereafter, Strategy 1 aims to attain this level as quickly as possible and then maintain that level through annual culling. By contrast, Strategy 2, while also aiming to attain the 0.1/km^2 goals as quickly as possible, permits numbers to rise to 0.25 animals//km^2 before culling begins again to get feral animal numbers down to 0.1/km^2. *A priori*, it is expected that the latter strategy may be cheaper to employ as there are increasing marginal costs of culling as feral animal densities decline.

An analysis is, therefore, undertaken of:

- the relative costs and benefits of the two control strategies
- the implications of a go-stop policy (i.e. the implications of a program that does not continue over the full time horizon).

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[3] Although the potential benefits of control to native communities of flora and fauna may also be large they are not considered in this report due to the limited existing information regarding their economic values and the scope of the overall project.

[4] In the view of feral animal control experts, this is the feral animal density level that is compatible with the INRM Plan goal of no deterioration in the extent, condition and functionality of the native Territory environments in which the ferals are found.

- the degree to which higher aerial culling costs affect total costs
- the degree to which a more sophisticated population modelling approach would affect the model results

Sensitivity analyses are also carried out with regard to a range of variables, including the proportion of feral animals actually competing with livestock for feed resources, the discount rate and the time horizon.

Findings

Benefits of controlling feral animals

In the context of this study, the principal economic benefits of controlling feral animals arise from the forgone (private) income from cattle production that it is avoided as a result of undertaking a given control strategy. The magnitude of the production loss avoided depends on a number of factors. These include:

- the current feral animal population. This is based on the most recent census data that is available and varies in future years according to natural growth rates and culling efforts.
- the proportion of the total feral animal population that are found on pastoral stations. This is estimated as 20% in all regions, except for horses and buffalo in the more highly developed Darwin Region, where the figure used is 30%.
- the proportion of pastoral properties that provides good grazing and where competition with feral animals actually takes place. This is estimated as 62%.
- the degree to which different feral animal species are considered to compete with cattle for limited grazing resources. The degree of competition is expressed as a proportion of the feed requirements of a given feral animal species relative to cattle. These are assumed to be as follows:
    - Buffalo/cattle = 1.5
    - Donkey/cattle = 1.2
    - Camel/cattle = 1.5
    - Horse/cattle = 1
    - Rabbit/cattle = 0.0083

The overall cost of feral animals to pastoralists is thus arrived at by applying these multipliers to the net income forgone p.a. from each head of cattle. The latter is conservatively estimated at $200 per head/year[5].

Costs of feral animal control

For all of the large mammals, the preferred method of control is shooting by helicopter.[6] Based on NRETA data collected from actual control activities of this type, it is recognised that there are increasing marginal costs as feral animal densities decline. This is because the labour and helicopter time required to shoot individual

[5] For example, of a hypothetical population of 200,000 camels, 40,000 (200,000 x 0.2) might be found on pastoral stations. Of these, only 24,800 (40,000 x 0.62) would actually be competing with cattle for grazing resources and this would be equivalent to running an additional 37,200 (24,800 x 1.5) cattle on those properties. This would be equivalent to a loss of cattle production equal to $7.44m p.a. (37,200 x $200).

[6] In so far as alternative methods (e.g. ground shooting) might be cheaper in some instances, the resulting control cost estimates should be interpreted as establishing an upper-bound.

animals increases as fewer target animals can be identified. Thus, at densities of over 0.25 animals/km^2, helicopter time is $13.50/animal and the associated labour (2 shooters plus pilot) is $1.40/animal. At densities of 0.11 – 0.24 animals/ km^2, these costs are $35 and $3.63, respectively.[7] Ammunition costs $1.50 per animal regardless of density.

Total control costs for each individual species are thus dependent on densities and hence regional population differences. We now examine these in detail.

Economics of camel control in the Central region:

Based on the results of the 2000 census and an assumed natural growth rate of 10% p.a., by 2007 (year 1 of our analysis) the estimated camel population in the NT portion of Central Australia (262,000 km^2) was 212,587. In the absence of control the population will increase to 390,000 head before stabilising as the maximum capacity of the resource base to support camels is reached. Such an equilibrium would be reached by the beginning of year 8 (2014).

Where control strategy 1 is employed, as can be seen in Table D1, the initial density of 0.81 animals/km^2 means that approximately 147,000 camels need to be removed (almost 70% of current population). Taking into account the natural growth rate of camels (10% p.a.), the population at the beginning of the second year will have reached approximately 72,000 head and a further 45,850 camels need to be removed to reach the target density of 0.1. Total population will then be 26,200 camels and the 10% natural increase of 2,620 camels will need to be removed each year thereafter.

Table 1 also presents the control costs that occur in each year under an annual camel culling strategy. Costs are made up of helicopter time (rental charges if helicopters belong to contractors or charges for fuel, maintenance and depreciation if owned by the agency in charge of the culling program), labour (shooters and pilots), ammunition, expenses associated with a monitoring program and overall project management. With the exception of the latter two items, all these costs are dependent on the number of camels being culled in any given year.

We note, however, that the stream of future costs cannot simply be added together as the value of a dollar spent in year 1 is not the same as the value of a dollar spent in later years. Future costs must be suitably "discounted" before they can be presented in present value terms. Using a typical 5% discount rate and assuming a 20 year time horizon[8], it can be seen that a camel control program would cost approximately $4.53m, which is equivalent to an annual present cost of approximately $363,500.

A similar analysis under a density sensitive control strategy (2) reveals total present costs of approximately $4.75m over 20 years, which is equivalent to an annualised present cost of approximately $380,800.

[7] Below a density of 0.1 animals/ km^2. these figures are $100 and 10.37, respectively. However, as our lowest target density is 0.1/ km^2 these costs do not actually play a role in the current model.

[8] We subject these assumptions to a sensitivity analysis in Section V.

In the case of camels, strategy 2 control costs are marginally higher overall even though culling is only carried out in years 1, 2, 8, 14 and 20 when camel densities have risen over 0.25. In years when the cull is carried out approximately 52,000 camels would have to be removed, compared to 2,620 p.a. under Strategy 1.

The direct economic benefits of camel culling are related to the potential for permitting increased cattle numbers or off-take. Table D1 shows that such benefits under the annual culling strategy are in the order of $55m over 20 years, which is equivalent to approximately $4.41m p.a. As can be seen in Table D1, these benefits far outweigh the costs of the control program, resulting in a benefit/cost ratio of more than 12.

The calculation of net benefits under strategy 2 reveals that strategy 1 is preferred as strategy 2 generates a lower level of net benefits. These are equivalent to a total of approximately $51.5m or $4.13m p.a.

Table D1. Camel Population (Central NT) and Control Costs under an Annual Culling Strategy (1)

	Year 1	Year 2	Year 3 and thereafter
Camel population including natural growth	212,587	72,050	28,820
Total area (km^2)	262,000	262,000	262,000
Camel density/ km^2	0.81	0.28	0.11
Target density/ km^2	0.25	0.10	0.10
Camels to be removed to achieve target (head)	147,087	45,850	2,620
Total animals remaining at end of year after culling	**65,500**	**26,200**	**26,200**
Control Costs (Strategy 1: annual culling)			
Cost of helicopter shooting	1,985,679	618,975	91,700
Ammunition	220,631	68,775	3,930
Labour	205,922	64,190	9,510
Status Monitoring	30,000	30,000	30,000
Mangement of control program	5,000	5,000	5,000
Total control cost in each year (Aus$)	**2,447,232**	**786,940**	**140,140**
Total Present Costs over 20 years at a 5% discount rate	**4,530,347**		
Annualised Present Costs based on 20 year time horizon and a 5% discount rate	**363,527**		
Total Present Benefits over 20 years at a 5% discount rate	**55,001,355**		
Annualised Present Benefits based on 20 year time horizon and a 5% discount rate	**4,413,451**		
Benefit/Cost Ratio	**12.14**		

Economics of horse and donkey control in the Central region:

The estimated population in the NT portion of Central Australia (240,000 km^2) was 2,000 donkeys and 18,000 horses in 2007. Without control the population of both species can be expected to increase to 240,000 head before stabilising as the maximum carrying capacity of the land is reached in year 15 (2021). Due to a higher natural population growth rate (22% p.a. for donkeys compared to 20% p.a. for horses), the proportion of donkeys to horses will also continuously increase over the time horizon under consideration[9].

Where control strategy 1 is employed, the initial density of horses (0.08/km^2) and donkeys (0.01/km^2) means that no animals need to be removed within the first two years (see Table 2). Taking into account the natural growth rate of horses (20% p.a.), the population at the beginning of the third year (2009) will have reached approximately 25,920, which is above the target density and hence 1,920 horses will need to be removed. 4,800 horses will need to be removed each year thereafter. The donkey population will reach a population above the target density only at the beginning of year 14 (2020), when the population will reach 26,528 donkeys, which corresponds to a density of 0.11 animals//km^2. After removing 2,528 donkeys in that year, 5,280 donkeys will need to be removed in year 15 and thereafter.

As shown in Table D2, the present costs of a horse and donkey control program in the Central region of NT would be approximately \$2.97m, which is equivalent to an annualised present cost of approximately \$238,500.

A similar analysis under a density sensitive control strategy (Strategy 2) reveals that it would be necessary to cull approximately 40,500 horses in years 8, 14 and 19, when their densities exceed 0.25 animals/km^2. Culling of donkeys would only be carried out in years 14 and 19. Total present costs of this strategy are approximately \$2.16m over 20 years, which is equivalent to an annualised present cost of \$173,662. Strategy 2 control costs are thus marginally lower overall than for Strategy 1.

Although following a strategy based on density sensitive control (Strategy 2) seems to be slightly cheaper, the net benefits arising from the two strategies needs to be compared once the potential benefits to the cattle industry have been accounted for. The direct benefits of the potential increase in cattle numbers under the annual culling strategy are in the order of \$8.6m over 20 years (\$689,588 p.a). Similar calculations for strategy 2 reveal that the net present benefits over 20 years are only \$6.67m (\$535,508 annualized). Hence, strategy 1 (annual culling) will be preferred.

[9] Over longer periods it might be expected that the horse population would eventually stabilise in niche areas where they cannot be out-competed by donkeys.

Table D2. Horse & Donkey Population (Central NT) and Control Costs under an Annual Culling Strategy

	Year 1	Year 2	Year 3 and thereafter
Horse Donkey population including natural growth (head)	20,000	24,040	28,897
Total area (km^2)	240,000	240,000	240,000
Horse density (head/km^2)	0.08	0.09	0.11
Donkey density (head/km^2)	0.01	0.01	0.01
Target density (head/km^2)	0.25	0.10	0.10
Horses to be removed to achieve target (head)	0	0	1,920 (4,800 from year 4)
Donkeys to be removed to achieve target (head)	0	0	0 (5,280 from year 15)
Animals to be removed to achieve target (head)	0	0	1,920
Total animals remaining at end of year after culling (head)	**20,000**	**24,040**	**26,977**
Control Costs (Strategy 1: annual culling)			
Cost of helicopter shooting	0	0	67,200
Ammunition	0	0	2,880
Labour	0	0	6,969
Status Monitoring	30,000	30,000	30,000
Management of control program	5,000	5,000	5,000
Total control cost in each year (Aus$)	**35,000**	**35,000**	**112,049**
Total Present Costs over 20 years at a 5% discount rate ($)	**2,973,098**		
Annualised Present Costs based on 20 year time horizon and a 5% discount rate ($)	**238,569**		
Total Present Benefits over 20 years at a 5% discount rate ($)	**8,593,788**		
Annualised Present Benefits based on 20 year time horizon and a 5% discount rate ($)	**689,588**		
Net Present Benefits over 20 years at a 5% discount rate ($)	**5,620,690**		
Benefit/Cost Ratio	**1.62**		

Economics of horse, donkey and buffalo control in the Arnhem Region:

Based on the results of the 1998 census and an assumed natural growth rate of 22% p.a. for donkeys, 20% for horses and 15% for buffalos, the maximum capacity of the resource base (200,000 animals) was reached in 2005. Thereafter, horses and donkeys began to out-compete buffalo leading to an estimated animal population in the Arnhem Region (101,000 km^2) of 29,253 donkeys, 60,088 horses and 110,659 buffalo in 2007.

Where control strategy 1 is employed (see Table D3), the initial density of donkeys (0.29 animals/km^2), horses (0.59 animals/km^2) and buffalo (1.10 animals/km^2) means that approximately 4,000 donkeys, 34,800 horses and 85,400 buffalos need to be removed in the first year of control. In total, approximately 124,250 animals need to be removed in the first year, approximately 60,000 in the second year and 5,750 animals in year three and thereafter.

As can be seen in Table D3, the present costs of a control program in the Arnhem Region would be approximately \$5.82m[10], which is equivalent to an annualised present cost of approximately \$467,000.

A similar analysis under a density sensitive control strategy (2) reveals the need to have equivalent culling levels in years 1 and 2, followed by the culling of approximately 47,000 head in years 7, 12 and 17. Total present costs are approximately \$5.86m over 20 years, which is equivalent to an annualised present cost of approximately \$470,000. Strategy 2 control costs are thus marginally higher than those of Strategy 1.

No substantial cattle herds are kept in the Arnhem Region and hence no direct benefits related to the potential for increased cattle numbers occur. The control of these feral species in the Arnhem Region of course has other benefits (e.g. related to biodiversity impacts and cultural values) and although these may be significant, assessing such indirect impacts are beyond the scope of this study.

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[10] We note that this compares to approximately \$850 million that was spent on the Brucellosis Tuberculosis Eradication Campaign over 30 years. However, this cost included much more than just aerial shooting of buffalo in Arnhem Land (C. McMahon, pers. com.).

Table D3. Horse & Donkey & Buffalo Population (Arnhem Region) and Control Costs under an Annual Culling Strategy (1)

	Year 1	Year 2	Year 3 and thereafter
Horse, donkey and buffalo population including natural growth	200,000	200,000	200,000
Total area (km^2)	101,000	101,000	101,000
Horse density	0.59	0.30	0.12
Donkey density	0.29	0.31	0.12
Buffalo density	1.10	0.29	0.12
Target density (animals/km^2)	0.25	0.10	0.10
Horses to be removed to achieve target	34,838	20,200	2,020
Donkeys to be removed to achieve target	4,003	20,705	2,222
Buffalos to be removed to achieve target	85,409	18,938	1,515
Animals to be removed to achieve target	124,250	59,843	5,757
Total animals remaining at end of year after culling (head)	**75,750**	**30,300**	**30,300**
Control Costs (Strategy 1: annual culling)			
Cost of helicopter shooting	1,677,375	807,874	201,495
Ammunition	186,375	89,764	8,636
Labour	173,950	83,780	20,896
Status Monitoring	30,000	30,000	30,000
Management of control program	60,000	60,000	5,000
Total control cost in each year (Aus$)	**2,127,700**	**1,071,417**	**266,026**
Total Present Costs over 20 years at a 5% discount rate	**5,818,811**		
Annualised Present Costs based on 20 year time horizon and a 5% discount rate	**466,916**		
Total Present Benefits over 20 years at a 5% discount rate	**0**		

Economics of horse and donkey control in the Victoria River District (VRD) region:

The current estimated donkey population in the VRD region (120,000 km^2) is approximately 53,100 and 39,500 horses, totalling approximately 92,600 animals. In the absence of control, the total population will increase to 200,000 head in year 6 (2012) before stabilising. Thereafter, the total number of animals would remain unchanged, although the proportion of donkeys would increase due to their higher natural population growth rate.

As can be seen in Table D4, the initial density of 0.33 horses/km^2 and 0.44 donkeys/km^2 means that approximately 32,600 animals need to be removed in year 1, 48,600 in year 2 and 5,040 thereafter. The total present costs for control strategy 1 is approximately $3.91m, which is equivalent to an annualised present cost of approximately $314,100.

A similar analysis under a density sensitive control strategy (2) reveals that the total present costs are slightly lower under such a strategy. They are in the range of approximately $3.66m over 20 years, which is equivalent to an annualised present cost of approximately $293,400.

Where this strategy is applied, culling is the same as under Strategy 1 in years 1 and 2, with approximately 38,300 animals being culled in years 7, 12 and 17.

Benefits of horse and donkey control

The direct benefits under the annual culling strategy (1) equal approximately $49.3m over 20 years, which is equivalent to approximately $3.95m. Total net benefits are therefore approximately $45.4m, which is approximately $4m more than under Strategy 2.

Table D4. Horse & Donkey Population (VRD) and Control Costs under an Annual Culling Strategy (1)

	Year 1	Year 2	Year 3 and thereafter
Horse and donkey population including natural growth (head)	92,592	112,175	135,907
Total area (km^2)	120,000	120,000	120,000
Horse density (animals/km^2)	0.33	0.30	0.12
Donkey density (animals/km^2)	0.44	0.31	0.12
Target density (animals/km^2)	0.25	0.10	0.10
Horses to be removed to achieve target (head)	9,471	24,000	2,400
Donkeys to be removed to achieve target (head)	23,121	24,600	2,640
Animals to be removed to achieve target (head)	32,592	48,600	5,040
Total animals remaining at end of year after culling (head)	**60,000**	**24,000**	**24,000**
Control Costs (Strategy 1: annual culling) ($)			
Cost of helicopter shooting	439,992	656,100	176,400
Ammunition	48,888	72,900	7,560
Labour	45,629	68,040	18,293
Status Monitoring	30,000	30,000	30,000
Management of control program	60,000	60,000	5,000
Total control cost in each year (Aus$)	**624,509**	**887,040**	**237,253**
Total Present Costs over 20 years at a 5% discount rate	**3,914,891**		
Annualised Present Costs based on 20 year time horizon and a 5% discount rate	**314,141**		
Total Present Benefits over 20 years at a 5% discount rate	**49,270,949**		
Annualised Present Benefits based on 20 year time horizon and a 5% discount rate	**3,953,628**		
Net Present Benefits over 20 years at a 5% discount rate	**45,356,057**		
Benefit/Cost Ratio	**3.90**		

Economics of horse, donkey and buffalo control in the Gulf region:

Based on the results of a 2000 census, the maximum population size for these 3 species in the Gulf region was reached in 2002. Thereafter, the proportion of horses and donkeys is estimated to have grown at the expense of buffalo. By 2007 the estimated population was 58,051 donkeys, 137,090 horses and 4,859 buffalos.

Where control strategy 1 is employed, the initial density of 0.98 (horses), 0.41 (donkeys) and 0.03 (buffalos) means that approximately 102,000 horses and 23,000 donkeys need to be culled in the first year of control in order to meet the year 1 target density of 0.25animals/km^2 (see Table D5). A further 56,700 would need to be culled in year 2 and 5,880 in years three to year eight. The buffalo population begins to require culling as of year 9, leading to total animals needing to be culled in year 9 as approximately 6,750 and 8,000 thereafter.

Control strategy 1 would cost approximately $6.31m, which is equivalent to an annualised present cost of approximately $506,500.

A similar analysis under a density sensitive control strategy (2) reveals that the total present costs are slightly lower under such a strategy. They are in the range of approximately $6.20m over 20 years, which is equivalent to an annualised present cost of approximately $497,700. Under this strategy, culling in the first two years would be identical to that under strategy 1, followed by the culling of approximately 45,000 animals in the seventh year and approximately 65,000 animals in years 12 and 17 is required.

The direct benefits under strategy 1 equal approximately $44.0m over 20 years, which is equivalent to approximately $3.53m p.a. As can be seen in Table 5, these benefits outweigh the costs of the control program, resulting in net benefits of approximately $37.7m and a benefit/cost ratio of 1.98.

Similar calculations for Strategy 2 reveal that the net present benefits are approximately $7m less than under Strategy 1.

Table D5. Horse, Donkey & Buffalo Population (Gulf Region) and Control Costs under an Annual Culling Strategy (1)

	Year 1	Year 2	Year 3 and thereafter
Horse, Donkey & Buffalo including natural growth (head)	200,000	90,288	40,306
Total area (km^2)	140,000	140,000	140,000
Horse density (animals/km^2)	0.98	0.30	0.12
Donkey density (animals/km^2)	0.41	0.31	0.12
Buffalo density (animals/km^2)	0.03	0.04	0.05
Target density (animals/km^2)	0.25	0.10	0.10
Horses to be removed to achieve target (head)	102,090	28,000	2,800
Donkeys to be removed to achieve target (head)	23,051	28,700	3,080
Buffalos to be removed to achieve target (head)	0	0	0 (2,100 from year 10)
Animals to be removed to achieve target (head)	125,141	56,700	5,880 – 7,980
Total animals remaining at end of year after culling (head)	**74,859**	**33,588**	**34,426 – 42,000**
Control Costs (Strategy 1: annual culling) ($)			
Cost of helicopter shooting	1,689,407	765,450	205,800
Ammunition	187,712	85,050	8,820
Labour	175,198	79,380	21,342
Status Monitoring	30,000	30,000	30,000
Management of control program	60,000	60,000	5,000
Total control cost in each year (Aus$)	**2,142,316**	**1,019,880**	**270,962 – 355,234**
Total Present Costs over 20 years at a 5% discount rate	**6,311,873**		
Annualised Present Costs based on 20 year time horizon and a 5% discount rate	**506,481**		
Total Present Benefits over 20 years at a 5% discount rate	**44,022,059**		
Annualised Present Benefits based on 20 year time horizon and a 5% discount rate	**3,532,444**		
Net Present Benefits over 20 years at a 5% discount rate	**37,710,186**		
Benefit/Cost Ratio	**1.98**		

Economics of horse and buffalo control in the Darwin region:

Based on the results of the 1998 census, the horse and buffalo population in the Darwin region (75,000 km^2) would have reached its maximum of 200,000 in 2004. Thereafter, horses would have out-competed buffalo so that by 2007 the estimated populations in the Darwin region were approximately 166,000 horses and 34,000 buffalos.

Where control strategy 1 is employed, as can be seen in Table D6, the initial density of 2.21 horses/km^2 and 0.45 buffalos/km^2 means that approximately 143,000 horses and 15,000 buffalos need to be removed in the first year and a further 15,000 horses and approximately 14,000 buffalos need to be removed in the second year. Thereafter, 1,500 horses and 1,125 buffalos will need to be removed each year.

Such a horse and buffalo control program in the Darwin region would cost approximately $4.58m, which is equivalent to an annualised present cost of approximately $367,200.

A similar analysis under a density sensitive control strategy (2) reveals that the total present costs are slightly lower under such a strategy. They are in the range of approximately $4.22m over 20 years, which is equivalent to an annualised present cost of approximately $338,700. An equivalent number of animals as under strategy 1 would be culled in years 1 and 2, with 29,800 animals having to be removed in years 8, 14 and 20.

Unlike in the other regions considered, where only 20% of camels, horses, donkeys and buffalos are considered to compete for pasture with cattle, in the more developed Darwin Region 30% of feral horses and buffalo are considered to do so.

The direct benefits of potential increase in the carrying capacity for cattle under the annual culling strategy (1) equal approximately $52.0m over 20 years, which is equivalent to approximately $4.17m p.a. These benefits outweigh the present costs of the control program, resulting in a net present cost of approximately $47.4m. A similar assessment of Strategy 2 reveals that net present benefits would be approximately $3.65m less.

Table D6. Horse & Buffalo Population (Darwin Region) and Control Costs under an Annual Culling Strategy (1)

	Year 1	Year 2	Year 3 and thereafter
Horse and buffalo population including natural growth (head)	200,000	44,063	17,625
Total area (km^2)	75,000	75,000	75,000
Horse density (animals/km^2)	2.21	0.30	0.12
Buffalo density (animals/km^2)	0.45	0.29	0.12
Target density (animals/km^2)	0.25	0.10	0.10
Horses to be removed to achieve target (head)	147,289	15,000	1,500
Buffalo to be removed to achieve target (head)	15,211	14,063	1,125
Animals to be removed to achieve target (head)	162,500	29,063	2,625
Total animals remaining at end of year after culling (head)	**37,500**	**15,000**	**15,000**
Control Costs (Strategy 1: annual culling) ($)			
Cost of helicopter shooting	2,193,750	392,344	91,875
Ammunition	243,750	43,594	3,938
Labour	227,500	40,688	9,528
Status Monitoring	30,000	30,000	30,000
Management of control program	60,000	5,000	5,000
Total control cost in each year (Aus$)	**2,755,000**	**511,625**	**140,340**
Total Present Costs over 20 years at a 5% discount rate	**4,575,868**		
Annualised Present Costs based on 20 year time horizon and a 5% discount rate	**367,180**		
Total Present Benefits over 20 years at a 5% discount rate	**51,966,025**		
Annualised Present Benefits based on 20 year time horizon and a 5% discount rate	**4,169,888**		
Net Present Benefits over 20 years at a 5% discount rate	**47,390,156**		
Benefit/Cost Ratio	**4.43**		

Economics of rabbit control in the Central region:

Rabbits currently occupy about 285,000 km^2 of central Australia. Based on mapping of rabbit populations at pre-disease levels, about ten percent of this area is highly suitable habitat with a current density of 2 rabbits/km^2. The density of rabbits in remaining 90% of their range is 0.2/km^2.

The rabbit population in 2007 was estimated to be 170,000[11]. Based on mapping of rabbit populations at pre-disease levels, priority areas for control comprise approximately 400,000 km^2 in the Central region of the NT. In the absence of a control program, population growth is expected to be in the order of 1% p.a.

In contrast to the other species described so far, rabbit control is assumed to be carried out mechanically by ripping out warrens through the use of a bulldozer. Due to infrastructure and institutional constraints, rabbits on only 2,000 km^2 p.a. can be ripped. Associated costs are $2,000 per km^2 plus an annual cost of $30,000 for status monitoring and $5,000 in management costs. Annual ripping costs are thus $4,035,000 and it would take 200 years to fully cover the above area.

Assuming the rabbits are evenly spread across each 2,000 km^2 section, ripping will result in the culling of 4000 rabbits p.a. at a cost of approximately $1,008 per rabbit. Given that each rabbit may be considered to consume 0.83% of that of cattle, pasture gained p.a. from a culling program is only equivalent to 7 head of cattle. This estimate is based on very limited data and some tenuous assumptions, and we suggest that it should be viewed as requiring further consideration.

Hence, the justification for such an expensive control program is mainly to be found in the fact that if the rabbit population develops disease resistance, as they have in other parts of Australia, the population growth rate could reach 300% p.a. In this case, the impact on both pasture and environmental quality could be severe.

Assuming that the ripping constraints could be eliminated such that 20,000 km^2 could be cleared each year, rabbits could be largely eliminated within 20 years. Based on the above figures, this would however have a total present cost of almost $500m or $40m p.a.

Economics of pig control in the NT.

On mainland NT, a best-guess estimate is that there are approximately 500,000 feral pigs with a population growth rate of 0.25 (or 0.5) and a density of 6 animals/km^2 over the area in which they are found. We further assume that the goal is to reduce the density by 75%, with a first year target of 50% and aerial shooting costs of $30 per animal during the first year and $150[12] per animal thereafter. Based on such assumptions, total present costs are $82.35m over 20 years, which is equivalent to $6.6m p.a.

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[11] Note that this number is relatively small compared to the NT rabbit population in earlier decades. Such a difference is largely attributed to a number of diseases that have affected these populations.

[12] Costs are relatively high compared to large herbivores as pigs are much more difficult to spot from the air.

This very high cost can be put into context to some degree from an experience of baiting/trapping of pigs on Melville Island where the goal has been to eradicate a recently established population, estimated at 200 pigs. In that particular case, approximately $60,000 was spent on trapping and shooting pigs (ground and helicopter) during the first year and a further $40k in each of the following nine years as the pigs became increasingly trap shy. The total cost of such a program over the 10 years to achieve eradication can be calculated at $328,000 or approximately $1,650 per pig.

Economics of fox control in the Central region.

Unlike in other parts of Australia, foxes are not considered to have any direct impacts on production in the NT as there is no sheep farming and, consequently, no lambs to prey on. The main impact of foxes can, therefore, be considered to be in terms of biodiversity (e.g. through predation of native animals). While States such as Western Australia control foxes through aerial dispersal of bait laced with 1080 poison four to six times per year, concerns over the potential impact on dingos prevent such an approach being carried out in the NT. In addition to dingos being a protected species, inadvertent poisoning of dingos could also be considered undesirable as they contribute to the control of feral cat and fox populations.

The NT has consequently focussed on research related to the development of fox-specific bait dispensing devices and their deployment. Such devices, while still using 1080, do not permit dingos or crows to access the bait.

Assuming that such devices are only deployed in areas of high value biodiversity where foxes are a threat, it would be necessary to establish 4 management sites each of 25 km^2. Accounting for the cost of the devices, their deployment and travel to/from the management sites on a regular basis, the total cost would be in the region of $50,000 p.a. A further $25,000 p.a. would also need to be spent in order to monitor both the impact of the baiting program on the target species and on the threatened species that the program is aiming to protect. Hence, a limited fox control program for the NT, can be estimated to be in the order of $75,000 p.a. This is equivalent to $934,700 over 20 years.

Unfortunately this degree of control will not meet the INRM objectives, since it treats only 100 km^2 of area. A more ambitious program that would meet the INRM goals would be to treat 10% of all the areas affected by foxes (approximately 66,500 km^2). The 10% level is used because this is complementary to the objective of reserving 10% of all vegetation types in Protected Areas. This could be achieved using the same approach as the Western Shield Program in Western Australia, where 34,000 km^2 is treated using aerial and vehicle based delivery of 1080 baits at an annual cost of $1.3 million. The cost for the NT program would be $2.7 million p.a., or $33.6 million over 20 years.

Economics of control of wild dogs in the NT.

Wild dogs include dingos, feral dogs and their crosses. Wild dogs are considered to have some level of impact on livestock production, through attacks that reduce the value of cattle hides and damage tails, as well as through causing actual cattle mortality. However, the extent to which the latter is a significant problem is subject of some debate. Wild dogs may also be considered to contribute to controlling kangaroo numbers (thereby reducing competition for pasture with livestock), as it is estimated that the NT kangaroo population is only 20% of that on the other side of the dog fence. On balance, it may be that these two factors cancel out from the perspective of pastoralists.

The negative impact of wild dogs on biodiversity is thought to be minor, with only a few dogs per annum being eliminated from National Parks, mainly because of human safety concerns. In the NT the latter costs $1,000 p.a. At the same time, wild dogs may contribute positively to native biodiversity through feral animal control, as there is some evidence to suggest that they help control feral goats and rabbits.

Given the above and the fact that dingos are a protected species, the NTG carries out control (but not eradication) of wild dogs on pastoral lands to mitigate the impact on cattle production but such that “these control measures are consistent with conservation of the dingo” (G. Edwards, pers. comm.). This implies that control strategies involve a limit to the number of baits used and that baiting takes place only in strategic locations where cattle losses are occurring.

Across the pastoral leases that exist in the NT, the NTG currently spends in the order of $250,000 p.a., while pastoralists who supply bait and distribute it spend roughly the same amount. Wild dog control in the NT compatible with the INRM Plan is therefore approximately $300,000 p.a. This is equivalent to $3.7m over 20 years. Also, dingoes are not classed as feral animals and so the cost should really only be the cost of managing hybrid dogs and wild domestic dogs. The aim of such management would be twofold: mitigate damage to cattle and protect the genetic status of the dingo.

Economics of cat control in the NT.

Given that there is no way of consistently managing the threat of feral cats over large areas on the mainland, cat control is achieved through fencing off areas of high biodiversity value, such as around Uluru and Watarrka. Costs are approximately $200,000/km and each fence is 4-5 km long. Fences need to be replaced once every 20 years, while annual maintenance costs (includes repairs for damage caused by fire, floods and camels) are approximately 5% of the construction costs. Based on these assumptions, current cat control costs in the NT are in total present cost terms $1,575,000 over 20 years, which is equivalent to $126,500 p.a. However, this is based only on four enclosures each of approximately 1 km^2. As with the fox control, if a goal is set to protect 10% of the affected area (i.e. 133,000 km^2) from cats, and it is assumed that exclosures are 10x10 km, then the full cost would be $421 million p.a or $5.24 billion over 20 years. Note that this estimate may need ongoing revision to reflect changing (typically increasing) costs of fencing.

Economics of cane toad control in the NT.

Cane toad control is largely considered to be impossible to achieve on mainland NT given the sparse human population, the lack of an effective broad-scale control method and the toad’s distribution. About $1.5m has been spent over the last 6 years, largely on public education and localised reduction in toad numbers. This funding may also be expected to decline to near zero over the next few years.

Some rough estimates may be made regarding cane toad control on NT islands based on mitigating measures currently being undertaken by shipping companies. Such measures include the use of traps, fencing, cargo quarantine, monitoring and occasional eradication (but not from islands already invaded). We assume that the shipping company spends $15,000 to fence off its mainland loading facilities, that annual maintenance of the fence is 5% of the construction cost, fence life is 15 years, monitoring costs $1,200 per week (covering both the mainland and the island loading/unloading facilities) and eradication expenses of $250,000 are incurred once every 10 years[13]. In this case, the total present costs of control over 20 years at a 5% discount rate is $1.12m, which is equivalent to an annualised present cost of approximately $90,250.

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[13] For simplicity we assume the eradication event occurs in years 5 and 15

Discussion

<u>*Total costs of control*</u>

A summary of the above analyses (See Table D7) reveals that a control program for all the species considered in this report would have a total present cost of more than $5 billion over a 20 year time horizon and assuming a 5% discount rate. This is equivalent to an annualised present cost of about $470m.

Cats contribute 89% of this cost with rabbits also making substantial cost. Some of these costs are based on well established and broadscale methods, while those for cats, rabbits and pigs are based on extrapolation of methods used at very small scale. The problem is that there are no effective broadscale methods for these species, so until more cost effective solutions can be found, it seems either that we will have to provide massive investment in these programs or else accept more modest goals (essentially to control them over small areas). Hence, while highlighting the importance of accurately assessing the costs of control of these species, the remaining analyses carried out in this paper focuses exclusively on the large ruminants.

<u>Table D7</u>. Total Present Cost of All Feral Animal Control in the NT

Species	**Present Cost ($)**	**Percentage of Total Present Cost**
Camels*	4,530,347	0.08
Central Horses and Donkeys*	2,973,098	0.05
VRD Horses and Donkeys*	3,914,891	0.07
Arnhem Horses, Donkeys and Buffalo*	5,818,811	0.10
Gulf Horses, Donkeys and Buffalo*	6,311,873	0.11
Darwin Horses, Donkey & Buffalo*	4,575,868	0.08
Rabbits	498,924,591	8.47
Pigs	82,353,686	1.40
Foxes	33,600,000	0.57
Dogs	3,738,663	0.06
Island Cane Toads	1,124,625	0.02
Cats	5,240,000,000	89.00
Total Present Cost	**5,887,866,453**	100.0%

* Total present costs under Strategy 1

Costs of large ruminant control

The total costs of a strategy 1 (annual culling) control program for the all the large ruminants are approximately $28.1m over a 20 year time horizon and assuming a 5% discount rate. This is equivalent to an annualised present cost of $2.26m. Equivalent figures for a control program based on density-sensitive culling are $26.7m and $2.14m respectively. The proportion of costs to be incurred in each region is also presented for both strategies.

The slightly higher cost-effectiveness of Strategy 2 is related to the fact that control costs increase exponentially and, therefore, it is cheaper to only cull animals at higher densities (i.e. when their population reaches a density of 0.25/km^2 or slightly higher), rather than when they are always close to 0.1//km^2, as is the case under Strategy 1. Nevertheless, the difference between the two strategies is small and the fact that the control cost curve used in this analysis is discontinuous rather than smooth would have tended to favour strategy 2. Furthermore, as we will see in the next sub-section, the issue of the magnitude of the economic benefits associated with the different strategies plays a key role in strategy choice.

Table D8. Total Present Cost and Annualised Cost of Large Ruminant Animal Control in the NT

	Strategy 1	% of total	Strategy 2	% of total
Camels	4,530,347	16.1%	4,745,753	17.8%
Central Horses and Donkeys	2,973,098	10.6%	2,164,216	8.1%
VRD Horses and Donkeys	3,914,891	13.9%	3,656,226	13.7%
Arnhem Horses, Donkeys and Buffalos	5,818,811	20.7%	5,733,509	21.5%
Gulf Horses, Donkeys and Buffalo	6,311,873	22.4%	6,202,393	23.2%
Darwin Horses, Donkey & Buffalo	4,575,868	16.3%	4,220,520	15.8%
Total Present Cost	**28,124,888**	**100.0%**	**26,722,617**	**100.0%**
Annualised Present Cost	**2,256,814**		**2,144,292**	

Direct Economic Benefits of Large Ruminant Control

This study assumed that large ruminant feral animals compete directly with livestock for available pasture. Accounting for such competition from only 20% of feral animals and assuming that the removal of feral animals would allow for higher cattle numbers and off-take, led us to identify potentially large benefits associated with feral animal control.

As noted previously, the total present cost of large ruminant control over 20 years under strategy 1 is $28.1m. This compares with over $208.8m in direct economic benefits to the cattle industry, resulting in net present benefits of a control program being approximately $180.7m. This compares favourably with the net benefits generated under Strategy 2 of $162.5m. Hence, although allowing animal densities to grow to somewhat larger levels under Strategy 2 before culling is resumed saves on control costs, the loss of pasture due to the increased number of animals between culls outweighs this benefit by $18.2m over 20 years. Accordingly we note that, with the exception of the Arnhem Region where there are few managed cattle, strategy 1 will always be preferred. The remainder of this paper thus focuses exclusively on strategy 1 costs.

Finally, we also note that although we have costed only the direct economic benefits of feral animal control to the cattle industry, other significant benefits of control also exist and are likely to be higher under strategy 1 as well. This includes the benefits to biodiversity, cultural values and reduced infrastructure damage. Although the valuation of such benefits is beyond the scope of this study and we do recognise that such valuation can play an important role in informing feral animal control policy, we would nonetheless argue that given the high direct economic returns to feral animal control, the valuation of the indirect benefits is not critical to the current analysis.

An additional point worth noting is that although the model used in this analysis would suggest that a limited control budget should be allocated to those species and regions where the highest net benefits could be captured (i.e. in descending order of priority, Central Region camels, VRD, Darwin and Gulf horses, donkeys and buffalo), the fact that environmental and cultural values have not been taken into account complicates such matters. Furthermore, risk factors (e.g. a rabbit population explosion following the development of disease resistance) would also need to be taken into account.

Potential implications of an under-resourced control program

As noted in Table D8, the total present cost of large ruminant control under strategy 1 is $28.1m over 20 years. This can be expressed as $2.26m p.a., where this number represents the equivalent amount in present costs terms that would have to be spent each year over 20 years. Such a calculation is useful for putting the total cost figure into context. Given that the current operating budget of the NTG feral animal control program is in the region of $1m p.a. (G. Edwards, pers. com.), it is clear that a significant increase would be required for the INRM Plan goals to be achieved with regard to large ruminants alone. Furthermore, the annualised figure we have calculated is only really useful if one can assume that costs are indeed spread evenly over the time horizon under analysis. This is in fact not the case, as can be seen in Table D10 which shows that a very high proportion of funds is spent in the initial years. For example, with the exception of horses and donkeys in the Central Region, 50-75% of all funds are spent by year 5.

Table D10. Proportion of total funds spent during initial years of a 20 year control program.

Proportion of funds spent within (years)	Camels	Central Horses and Donkeys	VRD Horses and Donkeys	Arnhem Horses, Donkeys and Buffalo	Gulf Horses, Donkeys and Buffalo	Darwin Horses, Donkeys and Buffalo
1	51%	1%	15%	35%	32%	57%
2	67%	2%	36%	52%	47%	67%
3	70%	5%	41%	55%	51%	70%
5	75%	18%	51%	63%	58%	75%
10	79%	44%	71%	78%	73%	85%

Consequently, Table D11 shows that, with the exception of camels, if a control program were to cease after 5 years, then despite having spent a large proportion of the control budget, all large ruminant feral numbers would have recovered to present levels within 21 years.

Table D11. Years required to reach initial population size if control program ceases.

	Camels	Central Horses and Donkeys	VRD Horses and Donkeys	Arnhem Horses, Donkeys and Buffalo	Gulf Horses, Donkeys and Buffalo	Darwin Horses, Donkeys and Buffalo
cease after 1 year	14	2	4	7	7	12
cease after 2 years	25	2	10	13	12	18
cease after 3 years	26	2	11	14	13	19
cease after 5 years	28	2	13	16	15	21
cease after 10 years	33	2	18	21	19	26

Sensitivity Analysis

The control program costs and net benefit estimates derived above are entirely dependent on the data provided by the relevant experts in this field. In so far as the results obtained can provide useful “ball park” figures upon which policy recommendations and future research priorities can be defined, it is useful to assess the robustness of the above findings. As such it is worthwhile exploring the degree to which the model results are driven by and sensitive to particular assumptions.

Degree of competition between large ruminants and cattle

As noted in Table D10, the total net present benefits arising from large ruminant control are large ($180.7m over 20 years). This result was based on a number of assumptions regarding the current feral population size, the proportion of that population which can be found on pastoral properties (20%, except in the Darwin Region, where it is 30%), the proportion of pastoral properties where such competition actually impacts cattle significantly (62%), a measure of the amount of pasture consumed by each feral animal species relative to cattle (varies between 100%-150% depending on the species) and the value of cattle off-take.

Table D12 shows how the net present benefits over 20 years change depending on the proportion of the large ruminant population which can be found on those pastoral properties. Where that proportion would be only 10% (15% in the Darwin Region) of the total feral population (i.e. at 50% of our baseline assumption) the net present benefits would be reduced to $82.1m. Even where only 3% (4.5% in the Darwin Region) of large ruminants are expected to occur on pastoral properties (i.e. at 15% of our baseline assumption, the returns are still positive ($9.0m). It is, however, interesting to note that the net present benefits do become negative in the Central Region for horses and donkeys at such low levels of competition. In such a case, the important environmental and cultural benefits of control would have to be assessed in addition to the direct economic benefits to the pastoral industry that we have considered here.

Nevertheless, overall the model results seem to be robust with regard to the potential net benefits that can be generated through a control program.

Control costs

Control costs were assumed to increase in inverse proportion to the density of feral animals being culled. The costs used in this report are based on NTG experience with feral animal control. We note, however, that in our model although most of the culling carried out was done after the first few years at a density near 0.1 animals/km^2, the non-continuous nature of the cost function used meant that control costs (helicopter plus labour) were almost always in the region of $38.60 per animal plus ammunition (relevant to an animal density of 0.1 – 0.24/ km^2), rather than closer to the $110 per animal plus ammunition (relevant to animal densities below 0.1/km^2). It is therefore worth exploring the degree to which varying control costs might affect our overall findings.

Baylis and Yeomans (1989)[14] argued that control costs increase exponentially according to the following formula

$$C = 22.44 * D \wedge -0.673 \qquad [1]$$

where C is the cost per kill and D is the density/km^2.

[14] Bayliss, P. and Yeomans, KM. 1989. Distribution and Abundance of Feral Livestock in the 'Top End' of the Northern Territory (1985-86), and Their Relation to Population Control. Australian Wildlife Research 16(6) 651 - 676

Based on this model and taking into account that helicopter flying costs per hour appear to have increased substantially[15] and would thus now be in the region of \$92 - \$148 per animal killed at a density of 0.12 animals/km^2. This would be equivalent to a cost 2.5 to 4 times greater than we have currently modeled.

As can be seen in Table D13, accounting for higher aerial shooting costs increases the total present cost of the control program significantly. A three-fold increase in aerial shooting costs relative to our baseline estimate results in total present costs increasing by \$21.5m over 20 years (\$49.6m - \$28.1m). While our model's results are clearly sensitive to the assumptions made regarding actual aerial shooting costs, we note that given that the present direct benefits of large ruminant control were identified as \$208.85m in Table D9, such an increase would not undermine the argument that the net benefits to feral animal control are likely to be positive and very large. In fact, it would only be at extremely low levels of feral animal populations being found on pastoral properties (3% level in Table D12), with a more than doubling of control costs, that this finding would be overturned.

Population Growth Rates

The rate at which natural population growth of feral animals takes place is another important factor in determining total present control costs in the model. The annual natural population growth rates used were 10% for camels, 15% for buffalo, 20% for horses and 22% for donkeys. A limit to total population numbers was imposed in some areas to account for carrying capacities. For simplicity, such growth rates were assumed to be constant regardless of density, although it is recognized that population growth rates are likely to be higher at lower densities and along the "invasion front" of an expanding population.

While it is recognized that spatially explicit, habitat-based, density-dependent population models have been developed (e.g. for pigs, horses and buffalo in Kakadu National Park), developing such models for this analysis was beyond the scope of this project.

Instead, we note that the model findings appear to be robust even under much higher population growth rates. For example, a 50% increase in growth rates across all species relative to the baseline figures described above would lead to an increase in total present costs of only \$6.6m (from \$28.1m to \$36.7m - see Table D14).

In addition to the fact that modeling such a 50% increase might in some cases lead to projected species growth occurring beyond maximum population growths observed in practice (e.g. Rmax of buffalo is 17.5% - C. McMahon, *pers. comm.*), we also note that such an increase in control costs of \$6.6m would still be small relative to the estimated direct economic benefits of control.

[15] Bayliss estimated that helicopter costs in 1989 were approximately \$220 p.h which is equivalent to \$500 in 2007 dollars. Given that helicopter costs are currently approximately \$800, it appears that such costs have increased much faster than suggested by the ABS consumer price index for transport in general.

Table D14. Total present control costs over 20 years with different population growth rates.

	Baseline population growth Rate	50% increase in population growth rate
Camels	4,530,347	6,151,376
Central Horses & Donkey	2,973,098	5,029,627
VRD Horses & Donkey	3,914,891	5,080,831
Arnhem Horses & Donkey & Buffalo	5,818,811	7,150,620
Gulf Horses & Donkey & Buffalo	6,311,873	8,064,189
Darwin Horses & Donkey & Buffalo	4,575,868	5,183,129
Total Present Cost	**28,124,888**	**36,659,773**

Infrastructure constraints

Assuming that a helicopter has two shooters, each of which cull one animal every 6 seconds over a 6.5 hour day, it would be possible to cull 7,800 animals per day. The population numbers presented in Tables D1-6 for strategy 1 (annual culling) imply that approximately 591,600 large ruminants need to be culled in year 1, 240,000 in year 2, 23,800 – 31,350 in years 3-14 and 34,100 thereafter. Hence, 76 helicopter days would be required in year 1, 31 days in year 2 and between 3-4.5 days thereafter.

Even considering that culling rates would be slower at lower densities and helicopter days would be spent travelling to remote locations targeted for a control effort, the amount of helicopter time, shooters and pilots required for such a control strategy does not seem to be unfeasible.

Discount rates and time horizons

Finally, we acknowledge that the choice of discount rates and time horizons plays an important role in determining the magnitude of the overall figures. While the actual choice of a specific discount rate and time horizon is somewhat arbitrary, the use of a discount rate between 3-10% is common in many analyses and the use of a 20 year time seems reasonable given the long-term effort that needs to be put into feral animal control.

Use of a higher discount rate would reduce the total present costs (as future costs are more heavily discounted), while a lower discount rate would have the opposite effect. A sensitivity analysis reveals that, for example, the use of a 3% discount rate over a 20 year time horizon would increase total present costs to $31.9m, while a 10% rate would reduce it to $21.9m, relative to the $28.1m 5% baseline rate. The choice of discount rate is therefore unlikely to change our conclusions regarding the large positive nature of the net present benefits likely to occur from large ruminant feral animal control.

Use of a longer time horizon would increase total present costs but reduce the annualized measure. A sensitivity analysis reveals that, for example, use of a 50 year time horizon at a 5% discount rate would increase total present costs to \$37.3m, while the annualized present costs would be \$2.0m. The comparative baseline figures for a 20 year time horizon at a 5% discount rate are \$28.1m and \$2.26m, respectively (Table 8). It is therefore apparent that given the use of discounting, a large proportion of the total present costs are captured during the first few years of any given time horizon (in this case 75% of the 50 year horizon present costs are captured within the first 20 years). Once again, we can also conclude that the choice of time horizon will not play an important role in influencing the overall conclusions.

Table D9. Net present benefits of control

	Camels	Central Horses & Donkey	VRD Horses & Donkey	Arnhem Horses & Donkey & Buffalo	Gulf Horses & Donkey & Buffalo	Darwin Horses & Donkey & Buffalo	**Total**
Strategy 1 – present costs	4,530,347	2,973,098	3,914,891	5,818,811	6,311,873	4,575,868	**28,124,888**
Strategy 1 – present direct benefits	55,001,355	8,593,788	49,270,949	0	44,022,059	51,966,025	**208,854,177**
Strategy 1 – Net present benefits	50,471,008	5,620,690	45,356,058	-5,818,811	37,710,186	47,390,157	**180,729,289**
Strategy 2 – present costs	4,745,753	2,164,216	3,656,226	5,733,509	6,202,393	4,220,520	**26,722,617**
Strategy 2 – present direct benefits	51,518,682	6,673,611	45,384,071	0	37,300,971	48,312,219	**189,189,554**
Strategy 2 – Net present benefits	46,772,929	4,509,395	41,727,845	-5,733,509	31,098,578	44,091,699	**162,466,937**
Net Present Benefits Strategy 1 minus Strategy 2	**3,698,079**	**1,111,295**	**3,628,213**	**-85,302**	**6,611,608**	**3,298,458**	**18,262,352**
Biodiversity Benefits of Control	Significant	Significant	Significant	Significant	Significant	Significant	Significant
Cultural Benefits of Control	Significant	Significant	Significant	Significant	Significant	Significant	Significant

Table D12. Net present benefit ($) of large ruminant control program with varying degrees of competition with cattle

	Camels	Central Horses and Donkeys	VRD Horses and Donkeys	Gulf Horses, Donkeys and Buffalo	Darwin Horses, Donkeys and Buffalo	Total
Net Present Benefit (initial level of competition = 20%)	50,471,008	5,620,690	45,356,058	37,710,186	47,390,157	**180,729,289**
Net Present Benefit (50% of initial level of competition = 10%)	22,970,330	1,323,796	20,720,583	15,699,157	21,407,144	**82,121,010**
Net Present Benefit (15% level of initial competition = 3%)	3,719,856	- 1,684,030	3,475,751	291,436	3,219,035	**9,022,048**

Table D13. Total present control costs over 20 years for large ruminants with varying aerial shooting costs

Total Present Costs	Camels	Central Horses and Donkeys	VRD Horses and Donkeys	Arnhem Horses, Donkeys and Buffalo	Gulf Horses, Donkeys and Buffalo	Darwin Horses, Donkeys and Buffalo	Total	Cost increase relative to baseline
Baseline	4,530,347	2,973,098	3,914,891	5,818,811	6,311,873	4,575,868	**28,124,889**	
1.5 times	5,016,486	4,079,416	4,850,058	6,887,017	7,609,411	5,062,934	**33,505,322**	*5,380,433*
2 times	5,502,624	5,185,733	5,785,225	7,955,222	8,906,949	5,550,001	**38,885,754**	*10,760,865*
2.5 times	5,988,763	6,292,051	6,720,392	9,023,428	10,204,487	6,037,067	**44,266,187**	*16,141,298*
3 times	6,474,901	7,398,368	7,655,559	10,091,633	11,502,025	6,524,133	**49,646,620**	*21,521,730*
5 times	8,419,454	11,823,639	11,396,227	14,364,456	16,692,177	8,472,397	**71,168,350**	*43,043,461*

Appendix E: Pastoralism as a threatening process to biodiversity in the Northern Territory

1. Threat Scores

The 'extent' of pastoral landuse was calculated using GIS, by intersecting the current NT cadastre (NTLIS) with coverages of the vegetation types and regions used for this project. Tenure was simplified to four classes – Pastoral, Aboriginal, Reserves (including defence land), Other (including urban and peri-urban areas) (Fig. E1). It was assumed that all land currently under a pastoral tenure is actually used for pastoralism. Some land under aboriginal tenures is also used for pastoralism, but no coverage was available to indicate the spatial extent of this use. Therefore it was assumed that 10% of aboriginal tenures in each region/vegetation type combination were used for pastoralism, except in the Arnhem region, where there is minimal pastoral use. The calculated proportion of each vegetation type in each region used for pastoralism is shown in Table E1; this was converted to extent using the classes 0: <5%, 1: 5-10%, 2: 10-25%, 3: 25-50%, 4: 50-90% and 5: >90%, and this extent score was used for each of the asset tables. The coarse scale of vegetation mapping used means that the true extent of some important vegetation types is inadequately represented in the GIS analysis. In the case of pastoral impacts, this is particularly relevant to riparian vegetation, which occurs (usually as individually small areas) throughout most regions and vegetation types. Therefore it was assumed that the extent of this vegetation type was proportional to the total extent of pastoral lands in the region (rather than to the extent of the mapped riparian communities).

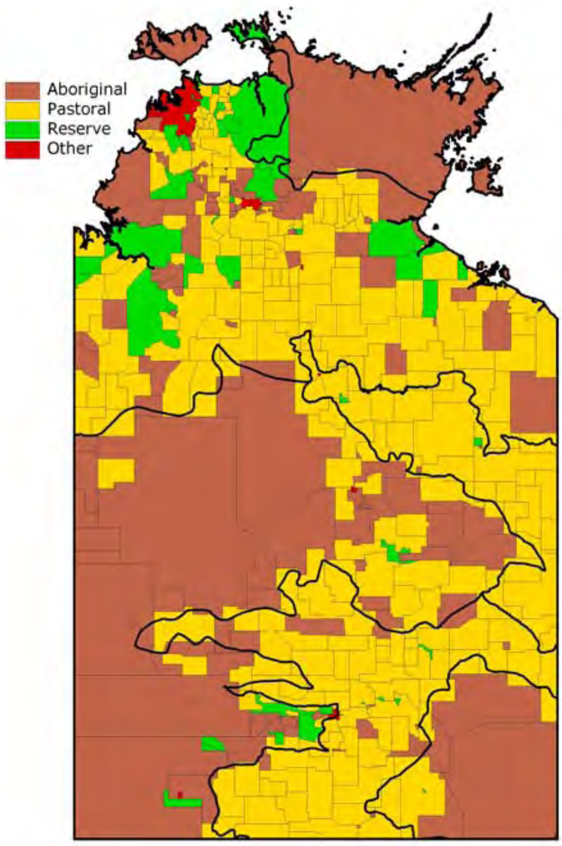

Figure E1. Simplified tenure in the NT used for 'extent' analysis, showing region boundaries

Table E1. Derivation of 'extent' scores for each vegetation type in each region. Table shows the percentage area within Aboriginal and Pastoral tenure of each vegetation type occurring in the region.

Vegetation type	Arnhem				Savanna			
	Area (km^2)	Abor.	Past.	Extent	Area	Abor.	Past.	Extent
Mangrove / coastal	1234	98.3%		0	917	20.7%	48.2%	4
Rainforest / riparian	768	73.4%		0	253	25.9%	29.7%	4
Floodplain	2606	99.5%		0	14859	36.4%	28.4%	3
Melaleuca	4237	99.6%		0	16654	25.3%	41.3%	3
Moist woodland	12294	89.0%	1.1%	0	69157	22.6%	47.1%	3
Euc forest	65116	94.5%	0.4%	0	69561	31.8%	41.2%	3
Euc woodland	14342	85.9%	0.2%	0	188440	18.4%	69.2%	4
"Heath"					4493	41.7%	34.0%	3
Acacia	442	100.0%		0	28490	14.5%	84.6%	4
Grassland					23315	11.7%	84.2%	4
Spinifex					13432	26.1%	73.9%	4
Bare								
Total		92.9%	0.5%			22.0%	60.2%	

Vegetation type	Barkly				Southern			
	Area (km^2)	Abor.	Past.	Extent	Area	Abor.	Past.	Extent
Mangrove / coastal								
Rainforest / riparian				5				5
Floodplain								
Melaleuca								
Moist woodland								
Euc forest								
Euc woodland	17078	0.0%	99.5%	5	9354	11.3%	86.7%	4
"Heath"	202		99.1%	5	6912	1.6%	98.4%	5
Acacia	11487		100.0%	5	95565	10.3%	89.4%	5
Grassland	62349	0.2%	99.2%	5	831		100.0%	5
Spinifex	1960	0.0%	97.3%	5	67400	9.6%	90.3%	5
Bare								
Total		0.2%	99.3%			9.7%	90.0%	

Vegetation type	Arid			
	Area (km^2)	Abor.	Past.	Extent
Mangrove / coastal				
Rainforest / riparian				3
Floodplain				
Melaleuca	1535	86.4%	13.6%	2
Moist woodland				
Euc forest				
Euc woodland	71443	49.7%	48.5%	4
"Heath"	5661	69.1%	30.6%	3
Acacia	39575	64.6%	31.8%	3
Grassland	2181	55.5%	44.5%	4
Spinifex	428292	83.6%	15.0%	2
Bare	2796	83.0%	17.0%	
Total		77.6%	20.8%	

Vegetation condition

Grazing by stock affects vegetation condition by modifying the structure and composition of the pasture layer (Landsberg *et al.* 1999). Typically this includes a reduction in the frequency and basal area of palatable perennial plants, and increase in unpalatable or 'weedy' species and increase in bare ground cover (Wilson 1990, James *et al.* 1999, Ash *et al.* 2001). In more extreme cases, grazing impacts may also affect the shrub and canopy layer, and there may also be a complex relationship between grazing and fire (so that, for example, fire is excluded and this promotes the growth of 'woody weeds' (Friedel *et al.* 1990)). Impacts of grazing on vegetation are not usually uniform within a vegetation type, but depend on the distribution of grazing pressure. Typically, vegetation is severely modified very close to watering points and relatively 'pristine' in water-remote zones (James *et al.* 1999, Ludwig *et al.* 1999).

Within the VAST framework adopted here (Table 6 of main report), most vegetation subject to pastoral use is in a "modified" state. Although the 'sacrifice zones' around waterpoints and some other areas of historically very high grazing pressure would fall into the "transformed" state, and there are also some pastoral areas that have been cleared and introduced pastures planted, these are small relative to the total regional extent of each vegetation type. Although pasture condition is monitored in the NT pastoral estate (Bastin *et al.* 1993, Karfs *et al*. 2000, NLWRA 2001), spatial mapping of vegetation condition is not adequate to objectively assess the condition of all vegetation types in each region. Therefore we assign a severity score of 1 to most vegetation types, and a score of 2 to vegetation types that are favoured for pastoralism and where grazing pressure is generally relatively high (floodplains, riparian vegetation, grasslands and acacia and chenopod communities in arid NT).

Sensitive and threatened species

There has been a substantial loss of biodiversity in Australian rangelands, which can at least be partly attributed to the impacts of pastoral landuse, and evidence suggests that there is continued decline and loss of species. The extent of these declines and the factors underlying them have been extensively reviewed and discussed (Burbidge & McKenzie 1989, Morton 1990, Reid & Fleming 1992, Lunney *et al.* 1994, Franklin 1999, Whitehead *et al.* 2001; Woinarski & Fisher 2003, McKenzie & Burbidge 2003). In this analysis, we are less concerned with species lost through unsustainable pastoral practices of the past and more with species that are currently declining but could be retained through investment in improved management of pastoral lands. Of the 187 extant species listed as threatened in the NT, pastoralism is considered a threat for 39 species (including 5 plants, 6 invertebrates, 1 frog, 2 reptiles, 12 birds and 13 mammals) (Woinarski *et al*. 2007; section 4 of main report), although for many of these species pastoralism is but one of a complex of threatening processes. Additionally, a large number of species are known to be "decreasers" (Wilson 1990) – that is, they decline in abundance with increasing grazing pressure (or decreasing distance from waterpoints). The proliferation of artificial waterpoint and intensification of pastoral use that aims to ensure the entirety of the pastoral estate is utilised by stock poses a particular threat to these species, as water-remote / minimally grazed areas are progressively reduced (Landsberg *et al.*1997, Biograze 2000).

Detailed studies have quantified the number of increaser and decreaser species for a range of taxa and in a number of environments within Australian rangelands, including mulga woodlands and chenopod shrublands (Landsberg *et al.* 1997), Mitchell and other grasslands on clay soils (Hoffman 2000, Fisher 2001, Churchill & Ludwig 2004) and tropical savanna woodlands (Woinarski & Ash 2002, Woinarski *et al.* 2002, Fisher & Kutt 2007). These studies suggest that between 10% and 40% of species in each major taxonomic group are likely to show a decreaser response to grazing pressure. Morever, decreaser species are likely to be species with relatively restricted habitat requirements (Fisher 2001), making them more susceptible to regional extinction. In some habitats particularly sensitive species may be found only at the most water-remote sites (Landsberg *et al.* 2002)

There is not a detailed account for each vegetation type in the NT of the number of species negatively affected by pastoral land use. We therefore assume that the results from the studies done to date can be generalised and that there is a set of "decreaser" species that continue to be negatively affected by pastoral landuse in all of the vegetation types (except mangrove/coastal communities, where there is minimal pastoral use), and most vegetation types were given a severity score of 1 for this asset (note that this is a conservative score, suggesting c. 10% of the biota is in decline due to pastoral use). The most mesic or run-on vegetation types (floodplains, riparian vegetation and melaleuca and chenopod communities in the arid zones), were given a severity score of 2 (10-25% in decline), as these types generally have high species richness and are subject to relatively high grazing pressure. For example, 32 of the 65 plant species of conservation significance occurring on the Barkly Tableland are associated with swamps, watercourses or waterholes (Fisher 2001).

Landscape function

The impacts of grazing livestock are not limited to effects on vegetation but also extend to other aspects of ecosystem function (Noble & Tongway 1983, Fleischner 1994, Ludwig et al. 1997, Ludwig et al. 1999). This includes changes in soil chemistry and structure through trampling and compaction, redistribution of nutrients through urine and faeces, disruption of microbial soil crusts and impacts on soil macrofauna and soil microbial biomass. Through vegetation removal and changes in soil surface condition, overgrazing may substantially alter the way water and nutrient are redistributed and lead to soil erosion. Degradation due to pastoral use is particularly focused in riparian zones and run-on areas (Pickup & Stafford-Smith 1993), which have historically been subject to very high stocking rates, and the 'riparian' vegetation type was given a severity score of 3 for this asset. Other vegetation types where grazing pressure is relatively high (floodplains, grasslands and acacia, melelauca and chenopod communities in arid NT) were given a severity score of 2, and most other vegetation types a score of 1. Mangrove communities and vegetation types in Arnhem Land were given a score of zero.

Production

We assume that there are no negative impacts of pastoral landuse on gross production and assigned a severity class of 0 to all cells in Table E2. In some cases, reduction of pasture condition by overgrazing may lead to long-term reduction in productivity, or land currently used for pastoralism could be more productively used for another landuse, but this was outside the scope of this analysis.

Cultural Values

The impact of pastoral landuse on cultural values is highly context-specific. To Aboriginal people displaced from their land and access to sacred sites and food resources the cultural impacts may be extreme. Many people, however, perceive the pastoral industry as a quintessential element of 'outback' Australia. This asset was therefore not assessed.

Table E2. Summary of 'severity' scores for three assets for each vegetation type in each region (AL=Arnhem Land, S=Savanna, B=Barkly, SA=Southern, A=Arid). Severity score for the "Production" asset was 0 in all cells, and the "Cultural" asset was not scored.

Vegetation type	Vegetation Condition					Species					Landscape Function				
	AL	***S***	***B***	***SA***	***A***	***AL***	***S***	***B***	***SA***	***A***	***AL***	***S***	***B***	***SA***	***A***
Mangrove / coastal	0	1				0	0				0	0			
Rainforest / riparian	0	2	2	2	2	0	2	2	2	2	0	3	3	3	3
Floodplain	0	2				0	2				0	2			
Melaleuca	0	1			2	0	1			2	0	2			2
Moist woodland	0	1				0	1				0	2			
Euc forest	0	1				0	1				0	1			
Euc woodland	0	1	1	1	1	0	1	1	1	1	0	1	1	1	1
"Heath"	0	1	1	2	2	0	1	1	2	2	0	1	1	2	2
Acacia	0	1	2	2	2	0	1	1	1	1	0	1	2	2	2
Grassland		2	2	2	2		1	1	1	1		2	2	2	2
Spinifex		1	1	1	1		1	1	1	1		1	1	1	1

2: References

Ash A, Corfield J & Ksiksi T (2001) *The Ecograze project: developing guidelines to better manage grazing country*. CSIRO Sustainable Ecosystems, Townsville.

Bastin GN, Sparrow AD and Pearce G (1993) Grazing gradients in central Australian rangelands: Ground verification of remote-sensing-based approaches. *Rangelands Journal* **15**, 217-233.

Biograze (2000). *Biograze: waterpoints and wildlife*. CSIRO, Alice Springs.

Burbidge AA and McKenzie NL (1989) Patterns in the modern decline of Western Australia's vertebrate fauna: causes and conservation implications. *Biological Conservation* **50**, 143-98.

Churchill TB and Ludwig JA (2004) Changes in spider assemblages along grassland and savanna grazing gradients in Northern Australia. *Rangelands Journal* **26**, 3-16.

Fisher A (2001). *Biogeography and conservation of Mitchell grasslands in northern Australia*. PhD thesis, Faculty of Science & Information Technology, Charles Darwin University, Darwin.

Fisher A and Kutt A (2007) *Biodiversity and land condition in tropical savanna rangelands: technical report*. Tropical Savanna Management CRC, Darwin.

Fleischner TL (1994) Ecological costs of livestock grazing in Western North America. *Conservation Biology* **8**, 629-644.

Franklin DC. (1999) Evidence of disarray amongst granivorous bird assemblages in the savannas of northern Australia, a region of sparse human settlement. *Biological Conservation* **90**, 53-68.

Friedel MH, Foran BD and Stafford-Smith DM (1990) Where the creeks run dry or ten feet high: pastoral management in arid Australia. *Proceedings of the Ecological Society of Australia* **16**, 185-194.

Hoffmann BJ (2000) Changes in ant species composition and community organisation along grazing gradients in semi-arid rangelands of the Northern Territory. *Rangelands Journal* **22**, 171-189.

James CD, Landsberg J and Morton SR (1999) Provision of watering points in the Australian arid zone: a review of effects on biota. *Journal of Arid Environments* **41**, 87-121.

Karfs R, Applegate R, Fisher R, Lynch D, Mullin D, Novelly P, Peel L, Richardson K, Thomas P and Wallace J (2000) *Regional land condition and trend assessment in tropical savannas*. Implementation Project Report, National Land and Water Resources Audit, Canberra.

Landsberg J, James CD, Morton SR, Hobbs T, Stol J, Drew A and Tongway H (1997) *The effects of artificial sources of water on rangeland biodiversity*. Environment Australia and CSIRO, Canberra.

Landsberg J, O'Connor TG and Freudenberger D (1999) The impacts of livestock grazing on biodiversity in natural systems, In *Vth International Symposium on the nutrition of herbivores,* (Eds, Jung, H.-JG and Fahey, GC) American Society of Animal Science, Savoy, Illinois.

Landsberg J, James CD, Maconochie J, Nicholls AO, Stol J and Tynan R (2002) Scale-related effects of grazing on native plant communities in an arid rangeland region of South Australia. *Journal of Appled Ecology* **39**, 427-444.

Ludwig JA, Tongway DJ, Freudenberger DO, Nobel JC and Hodgkinson KC (1997) *Landscape ecology, function and management: principles from Australia's rangelands.* CSIRO, Melbourne.

Ludwig JA, Eager RW, Williams RJ and Lowe LM (1999) Declines in vegetation patches, plant diversity, and grasshopper diversity near cattle watering-points in the Victoria River District, Northern Territory, *Rangeland Journal* **21**, 135-149.

Lunney D, Hand S, Reed P and Butcher D (eds) (1994) *Future of the fauna of western New South Wales*. Royal Zoological Society of New South Wales, Sydney.

McKenzie NL and Burbidge AA (2003) Mammals, In *Australian Terrestrial Biodiversity Assessment 2002,* National Land and Water Resources Audit, Department of Primary Industries and Energy and AGPS, Canberra. Pp. 83-96.

Morton SR (1990) The impact of European settlement on the vertebrate animals of arid Australia: a conceptual model. *Proceedings of the Ecological Society of Australia* **16**, 201-213.

Noble JC and Tongway DJ (1983) Herbivores in arid and semi-arid rangelands. In *Australian soils: the human impact,* (Eds JS Russell and Isbell RF), pp. 243-270. University of Queensland Press, St Lucia.

NLWRA (2001) *Rangelands – tracking changes. Australian Collaborative Rangeland Information System*. National Land and Water Resources Audit, Canberra.

Pickup G and Stafford-Smith DM (1993) Problems, prospects and procedures for assessing the sustainability of pastoral land management in arid Australia. *Journal of Biogeography* **20**, 471-487.

Reid J and Fleming M (1992) The conservation status of birds in arid Australia. *Rangelands Journal* **14**, 65-91.

Whitehead P, Woinarski JCZ, Fisher A, Fensham R & Beggs K (2001) *Developing an analytical framework for monitoring biodiversity in Australia's rangelands.* Report to the National Land and Water Resources Audit, Tropical Savannas CRC, Darwin.

Wilson AD (1990) The effect of grazing on Australian ecosystems. *Proceedings of the Ecological Society of Australia* **16**, 235-224.

Woinarski J, Pavey C, Kerrigan R, Cowie I and Ward S (2007) *Lost from our landscape: threatened species of the Northern Territory*. Northern Territory Department of Natural Resources Environment and the Arts, Darwin.

Woinarski JCZ and Fisher A (2003) Conservation and the maintenance of biodiversity in the rangelands. *The Rangeland Journal* **25**, 157-171.

Woinarski JCZ and Ash AJ (2002) Responses of vertebrates to pastoralism, military land use and landscape position in an Australian tropical savanna. *Austral Ecology* **27**, 311-323.

Woinarski JCZ, Andersen AN, Churchill TB and Ash AJ (2002) Response of ant and terrestrial spider assemblages to pastoral and military land use, and to landscape position, in a tropical savanna woodland in northern Australia. *Austral Ecology* **27**, 324-333.

Appendix F: Consideration of other factors that threaten biodiversity in the NT

In this report, we have compared, for five main threats (fire, feral animals, weeds, land clearing and pastoralism), the impacts upon biodiversity, and (for feral animals, fire and land clearing) the relative cost-effectiveness of management options. However, some native species and ecological communities are affected by other factors. This section provides a brief account of some of those factors, species and environments.

Climate change

Our main report did not provide particular focus on climate change. But climate change is likely to substantially exacerbate existing threats and/or provide a primary threat to some NT species and environments. However, unlike the threats described in the main report, it is difficult, if not futile, to consider management costs that would be associated with local- or regional-scale control of climate change. This is partly because climate change scenarios for the NT are still somewhat imprecise, but more because local- or regional-scale actions will not serve alone to control changing climate at those locations. Further, while it is possible to estimate management costs for such threatening factors as fire management, land clearing or control of feral animals, based on projections from current management actions, such is not the case for management to mitigate the impacts of climate change because there are no relevant current actions that are being undertaken.

Territory biodiversity will be affected by global climate change. It probably already is being affected, with relative increases over the last century in rainfall in northern Australia and increased atmospheric carbon contributing to a broad-scale tendency for vegetation to "thicken" (increase in woody basal area), in at least some regions of the NT. Climate change projections for the Northern Territory remain imprecise, but the most likely changes include increased frequency of severe cyclones, increased frequency of extremely hot days, increased severity or incidence of dry periods ("drought") in central Australia, rise in sea levels, warmer and more acid seas, and ongoing increases in concentration of greenhouse gases in the atmosphere (particularly elevated CO_2 levels) (e.g. Hennessy et al. 2004). Many changes may be more subtle but still ecologically significant (e.g. change in onset or duration of the wet season) or may result in compounded impacts in other threats. Many of these changes will directly affect Territory biodiversity. Some migratory Territory species will also be affected by even more marked climate changes elsewhere in the world.

To some extent, the biodiversity of the Northern Territory may be relatively well insulated against climate change, because (i) the Territory does not contain many habitats with very narrow and sharply-etched climatic delimiters (such as alpine areas), (ii) many Territory environments have had a turbulent recent (<20,000 years) history, marked by dynamism and the retention of more robust biodiversity elements; (iii) the Territory contains gradual climatic gradients and largely continuous natural landscapes, making it feasible for species to track suitable climatic conditions by dispersing to favourable areas (although, of course, the very gradualness of those existing climatic gradients means that species may have to move very large distances to re-position themselves); and (iv) some of the Territory's existing refugial areas (notably the deeply dissected sandstone plateau of western Arnhem Land) are likely to retain a substantial functionality as refugia.

Nonetheless, "relative" is a slippery word, and the rapidity and extent of human-induced climate change may have unprecedented adverse impacts on the Territory's biodiversity, ecosystem processes and environmental health. Some of these impacts can be predicted with reasonable confidence. Some of these impacts can be minimised with sufficient anticipatory actions.

The most likely major consequences of climate change for Territory biodiversity are:

(1) loss of the extensive coastal floodplain systems through sea level rise. These highly productive floodplains are the main breeding grounds for magpie geese and other waterfowl, and are important nursery areas for barramundi and other fish. These floodplain systems are particularly susceptible given their flatness, low elevation and proximity to the sea. Even minor changes in the salinity of their creek systems will result in very marked and extensive changes in vegetation. These floodplain systems cannot simply move inland ahead of the rising sea levels – the existing landforms will not allow that.
(2) loss or shrinkage of many Territory islands. In part because of their isolation, many existing Territory islands now support particular concentrations of threatened or narrowly endemic species, and many provide key nesting sites for seabirds and marine turtles.
(3) reduced viability of coral reef systems, because of changed water temperature regimes and/or increased acidity.
(4) increased stress on refuge areas in central Australia, and fewer or shorter periods of recovery times for plants and animals enduring through extended dry periods.
(5) possible problems for reptile species, such as many turtles, crocodiles and some lizards, for which temperature determines the sex of hatchlings.
(6) probable increases in the extent and severity of fires may further degrade some environments (such as sandstone heathlands) and further reduce the abundance of fire-sensitive species.
(7) decrease in hollow availability and forest structure in coastal areas, because of increased intensity of cyclones, will reduce habitat suitability for the large proportion of Top End animals that are dependent upon hollows for nesting or roosting.
(8) likelihood of new diseases, weeds or pests or increased incidence of some existing diseases, weeds or pests, that may impose new or increased pressures on Territory plants and animals.
(9) change in the location, or loss, of climatically suitable areas for some species or environments.

There are management, adaptation or interventionist options for some of these biodiversity consequences. These are summarised in the following tables, based on a subjective assessment of cost and feasibility (likelihood of success), however we caution that this is a very preliminary assessment and a more rigorous appraisal should be undertaken as part of a risk assessment and broader response to climate change for the NT. Nonetheless, for some of the likely impacts upon biodiversity of climate change, there may be no practical management responses.

Table F.1. Range of management actions that may reduce impacts of climate change on NT biodiversity. Note that each table represents one of the nine likely impacts described on the preceding page.

loss of floodplains		*likelihood of success*		
		low	medium	high
cost	low			
	medium		translocation of some elements to more inland lake systems	
	high	construction of a system of barrages to stem seawater intrusion	selection of one or few floodplains to intensively manage through a range of barrage and other mechanisms.	

loss or shrinkage of islands		*likelihood of success*		
		low	medium	high
cost	low			
	medium		translocation of some endemic or threatened biodiversity to less exposed areas	
	high			

reduced viability of coral reef systems		*likelihood of success*		
		low	medium	high
cost	low	no feasible local- or regional-scale management response		
	medium			
	high			

loss of refugial function		*likelihood of success*		
		low	medium	high
cost	low			
	medium		reduction in pressures from other current threatening processes (e.g. fire, water extraction, feral animals)	
	high			

temperature-related sex determination for some reptile species		*likelihood of success*		
		low	medium	high
cost	low			
	medium			
	high	large-scale artificial incubation in controlled climates; ex situ populations		

exacerbation of fire regimes		*likelihood of success*		
		low	medium	high
cost	low			
	medium		more intensive manipulation of fire regimes (particularly preventative burning)	
	high		reduction in extent and incidence of exotic grasses	

decrease in hollow availability and forest structure		*likelihood of success*		
		low	medium	high
cost	low			
	medium			
	high	broad-scale provision of artificial nest-boxes		

entry and spread of new diseases and other threatening factors		*likelihood of success*		
		low	medium	high
cost	low			
	medium			increase in surveillance, especially along the northern coastline
	high			

changing location (or loss) of climatically-suitable areas		*likelihood of success*		
		low	medium	high
cost	low			
	medium		maintain continuity in native vegetation	
	high		translocate species for which natural dispersal may be infeasible	

Note that many or most of these management actions have medium to high costs but low to medium likelihood of success, and that some possible impacts have no feasible ameliorative management responses.

Note also that some climate change impacts may not be readily predictable. Partly for this reason, and partly to allow flexibility in prioritisation of management responses and adaptive management generally, it is considered important that a broad-scale and broad-brush biodiversity monitoring program be established that may provide early warnings of changes in species' abundances or distribution in response to climate change.

Disease

Disease and pathogens can have major impacts upon biodiversity. For example, dieback caused by the cinnamon fungus *Phytophthora cinnamomi* is recognised as a Key Threatening Process under the *Environment Protection and Biodiversity Conservation Act*, causing the endangerment of many Australian native plant species. The devil facial tumour disease is a spectacular current example of disease leading to drastic decline in a native animal species, in this case, the Tasmanian devil. Chytrid fungus is another recent example, and has probably been a primary cause of the extinction of many frog species world-wide. Other diseases, such as hendra virus, scrub typhus and avian influenza, are also notable in affecting both wildlife and humans.

However, there is remarkably little information about the prevalence or seriousness of disease in NT biodiversity, the extent to which this incidence may be changing, and the feasibility and cost of management responses. In part, the lack of information possibly reflects a relatively low incidence for at least some conspicuous or high profile diseases. For example, the NT is thought to be the only Australian jurisdiction in which chytrid fungus is not present, and phytophthora-related dieback is very limited and highly localised in the Territory.

Nonetheless, disease has the potential to be a major threat for many Territory plant and animal species; and, as evident from the case of the devil facial tumour disease, well-considered management may be critical for conservation outcomes. Some diseases are likely to newly arrive in or increase in the NT because climate change may encourage the spread of their vectors or make NT environments more suitable for them.

Management priorities should be:

- establishment of a broad-scale program for establishing a baseline incidence of disease (and its impacts) across a widely representative range of NT biodiversity;
- establishment of a broad-scale monitoring program to detect changes from this baseline;
- establishment of a sensitive surveillance program that is capable of providing early warning of new disease outbreaks or incursions, with this program informed by risk-assessment of likely new diseases, their potential impacts and entry points;
- consolidation of existing quarantine standards and programs, aimed at reducing the risk of entry of new diseases;
- development of a tool-box of procedures that provide the most cost-effective and efficient controls to diseases that have substantial detrimental impacts to biodiversity;
- appropriate application of management responses, especially for new incursions or outbreaks.

Non-native invertebrates

The main body of this report considers threats posed by a range of non-native vertebrates, and by non-native plants. There is another group of exotic species whose occurrence and impacts are typically less conspicuous: non-native invertebrates. With the exception of a few species, little is known of the distribution, ecology or impacts of these species in the NT.

But such impacts may be substantial. The most high profile invertebrate pests in the NT are invasive (“tramp”) ants, particularly the big-headed ant *Pheidole megacephala* and the yellow crazy ant *Anoplolepis gracilipes*. Both of these species can have “a devastating impact on native invertebrates” (Andersen 2000), but may also detrimentally affect vegetation and ground-dwelling vertebrates. In the NT, the yellow crazy ant is largely restricted to north-eastern Arnhem Land. Intensive baiting has led to some at least localised eradication, for example in parts of Kakadu National Park.

The European honey bee *Apis mellifera* is now widespread in the NT. It may compete for food and shelter resources with native bees and reduce the availability of hollows to the many native vertebrate species that are dependent upon them.

Non-native invertebrates may pose a major hazard to aquatic and marine systems. The most celebrated example in the NT is of the black-striped mussel *Mytilopsis sallei*, a species capable of transforming marine ecosystems. A rapid management response averted the establishment of this species in Darwin Harbour in 1999.

Non-native invertebrates (such as some Asian mosquito species) may also act as vectors for the spread of novel diseases, capable of affecting wildlife and humans.

Management priorities should be:

- establishment of a sensitive surveillance program that is capable of providing early warning of new outbreaks or incursions of high priority non-native invertebrates, with this program informed by risk-assessment of likely new colonists, their potential impacts and entry points;
- consolidation of existing quarantine standards and programs, aimed at reducing the risk of entry of new non-native invertebrates;
- development of a tool-box of procedures that provide the most cost-effective and efficient controls to non-native invertebrates that have substantial detrimental impacts to biodiversity;
- appropriate application of management responses, especially for new incursions or outbreaks, and for the eradication of high-impact non-native invertebrate species in sites with high biodiversity conservation values..

“Other” non-native vertebrates

The main body of this report considered impacts and management options for a series of non-native vertebrate species, comprising donkey, horse, camel, water buffalo, pig, fox, cat, wild dog, rabbit and cane toad. This is a substantial list, but there are additional species that have some impacts on NT biodiversity. Other non-native vertebrates present in the NT include the Asian house gecko *Hemidactylus frenatus*, flowerpot blind snake *Ramphotyphlops braminus*, tree sparrow *Passer montanus*, rock dove *Columba livia*, spotted turtledove *Streptopelia chinensis*, goat *Capra hircus*, house mouse *Mus musculus*, black rat *Rattus rattus*, and gambusia (“mosquito fish”) *Gambusia holbrooki*.

Of these species, the Asian house gecko appears to be expanding at the expense of native geckoes (*Gehyra* species), the flowerpot blind snake has no known impacts; the tree sparrow and doves may possibly spread exotic diseases to native wildlife and may compete with them for food and nest sites; the goat is currently limited to a small set of islands (where it may have major impacts on vegetation); and the house mouse is widespread in central Australia, but its impacts are not well determined. The black rat is extremely abundant in Darwin suburbs and the peri-urban area and on some islands, and it has been recorded increasingly frequently from a range of sites remote from infrastructure in the Top End. Particularly on islands it is recognised as a major threat to biodiversity. It may spread exotic diseases to native wildlife, compete directly with native rodents, and predate on native wildlife (including the eggs and young of birds).

The NT's aquatic systems are notable for their rich and intact communities of fish and other aquatic fauna. Some exotic fish species may threaten these communities. Infestations of Gambusia are currently highly localised in the Alice Springs region, but this small exotic fish has elsewhere in the world (including in most other Australian states) proven to be a highly effective invasive colonist with substantial impacts on native aquatic fauna. The Tilapia (spotted mangrove cichlid or spotted tilapia) *Tilapia mariae* is now widespread in Queensland rivers, and notable for rapidly building large populations.

The NT NRETAS is currently preparing a *Northern Territory Vertebrate Pest Animal Strategy*, that will provide management priorities and guidelines for many of these species. Broadly, the management priorities should be comparable to those outlined above for invasive invertebrate species, that is:

- establishment of a sensitive surveillance program that is capable of providing early warning of new outbreaks or incursions of high priority non-native vertebrate pests, with this program informed by risk-assessment of likely new colonists, their potential impacts and entry points;
- consolidation of existing quarantine standards and programs, aimed at reducing the risk of entry of new non-native vertebrates;
- development of a tool-box of procedures that provide the most cost-effective and efficient controls to non-native vertebrates that have substantial detrimental impacts to biodiversity;
- appropriate application of management responses, especially for new incursions or outbreaks, and for the eradication of high-impact non-native vertebrate species in sites with high biodiversity conservation values..

Exploitative use

Some NT native animal and plant species are subject to commercial, recreational or customary use. Commercially and recreationally exploited species include many marine and freshwater fish species, a range of prawn species, mudcrabs, waterfowl, crocodiles, some reptiles used in the pet trade, ironwood, cycads, Kakadu plum, and quandong. Commercial trade in Indigenous artefacts includes use of hollow-bearing trees for didgeridoos, woollybutts for bark paintings, pandanus for weaving, and kapok and ironwood for carving.

The more substantial or intensive of these enterprises are typically regulated through management plans, assessment of populations, establishment of quotas, and assessments of sustainability. In some cases, such as for prawn fisheries, these commercial industries may also have impacts on non-target species (e.g. marine turtles and sea-snakes), and these impacts are typically considered (and actions taken to mitigate them) in management planning and assessment of sustainability.

Changed hydrology

There is rapidly increasing use of water in many areas of the NT, through proliferation of bores taking groundwater (for livestock, horticulture, mining or human consumption), and (less so) direct extraction of waters from rivers, and impoundment. A range of NT vegetation communities (such as spring-fed rainforests) are "groundwater-dependent ecosystems". Their persistence will require regulation that provides for adequate water to maintain environmental services. Particular water regimes (dry season flows, flooding regimes, connectedness) may also be critical for the persistence and breeding of many aquatic animal species, such as pig-nosed turtles, magpie geese, crocodiles and barramundi, and extraction of water or impoundment may so modify these regimes that habitat suitability for these species is compromised.

The biodiversity in aquatic systems may also be affected by sedimentation or pollution arising from surrounding land-use. Historically, poorly regulated mining ventures in the NT (most notably the Rum Jungle uranium mine) resulted in heavy metal pollution that had catastrophic impacts on aquatic biodiversity for many kilometres downstream. Such excesses and poor practice are unlikely now, but there remains some potential for developments to affect water quality and flow regimes, and thereby detrimentally affect some aquatic biodiversity.

Tourism

Many tourists visit the NT, often with particular interest in wildlife. It is conceivable that over-use of some sites by tourists may have localised impacts on biodiversity, such as through trampling, disturbance, pollution, changes in the behaviour of wildlife, and spread of exotic organisms. Elsewhere in the world, tourism has particularly threatened some highly localised sites where disturbance may have major impacts (such as some island seabird colonies). However, there is little evidence to suggest any substantial detrimental impacts of tourism on NT biodiversity. One possible exception may be the presumed loss of one of the then three known sites of the highly localised and threatened desert sand-skipper *Croitana aestiva* (a butterfly) through the development of a car park for tourism at Standley Chasm (http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=26238).

The occurrence and severity of different threatening factors

The text above describes a range of factors that threaten some components of NT biodiversity. We recognise that there may be further factors that have some localised or idiosyncratic impacts upon some NT native plant or animal species; but we suggest that the threats listed are those of most concern.

To provide a simple illustration of the distribution and severity of threats, we amplified the assessment provided in Section 4 of the main report, that considered listed threats to all NT threatened plants and animals. We cross-tabulated each listed threatened plant and animal species, its distribution for all NT IBRA sub-regions (Thackway and Cresswell 1995), and its listed threats. We then tallied the number of species in each subregion against each threat type. As a simplification, we assumed that the threats listed for any species operated in every subregion in which it occurred: this simplification resulted in some anomalies, such as considering that cane toads affected some species in arid sub-regions (because some native species affected by cane toads occurred in those subregions even though cane toads did not.

For each threat factor, we present four variations on these tallies:

- A is simply a tally of the number of threatened species per subregion affected by the given threat;
- B is as for A, but excludes fish and extinct species;
- C is as for B, but weighted by the proportion of subregion/NT records for each species (i.e. if relatively few of the NT records for a listed threatened species occur in a given subregion, then the score for that subregion will be releatively low); and
- D is as for B, but weighted by the number of threats listed for any given threatened species (e.g. if fire was the only listed threat for a given species, it would score 1, but if three additional threats were listed for that species fire would score 0.25).

The resulting maps are given in Figure F1. With the caveat that these are based only on threatened species, these maps provide an insightful indication of the relative impacts of different threats across the NT, and of the relative distribution of each threat. These maps confirm the relative importance of feral herbivores, feral predators, pastoralism, fire (especially) and weeds as the major threats to NT biodiversity. They also suggest that different subregions have different constellations of major threats, with the ordering of threats varying substantially between different areas.

References

Andersen AN (2000) *The ants of northern Australia: a guide to the monsoonal fauna*. CSIRO, Melbourne.

Hennessy K, Page C, McInnes K, Walsh K, Pittock B, Bathols J & Suppiah R (2004) *Climate change in the Northern Territory*. CSIRO, Melbourne.

Thackway R & Cresswell I (1995) *An interim biogeographic regionalisation for Australia: a framework for setting priorities in the National Reserves Cooperative Program*. Australian Nature Conservation Agency, Canberra.

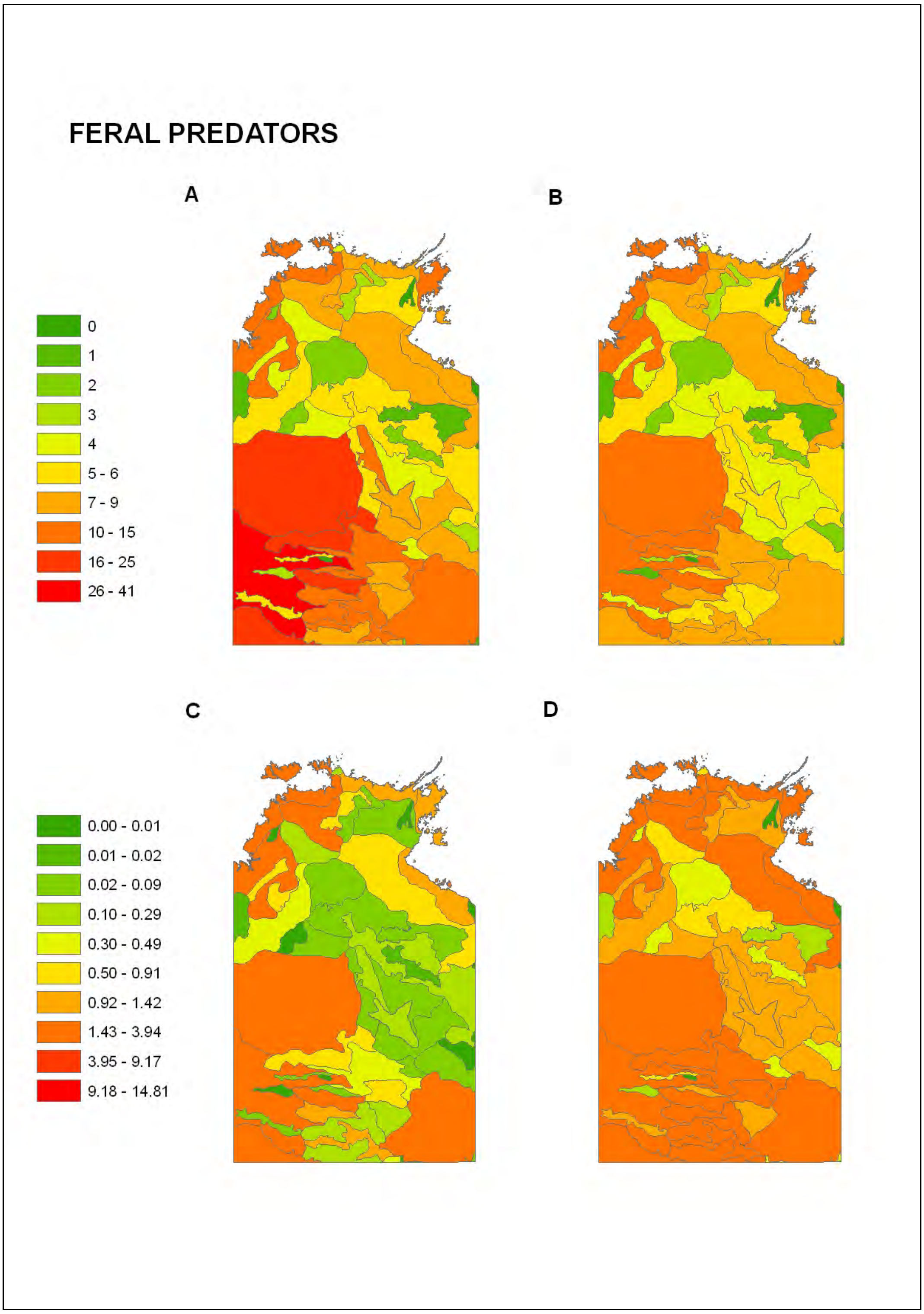

Figure F1.1. Relative impacts of feral predators on NT threatened species.

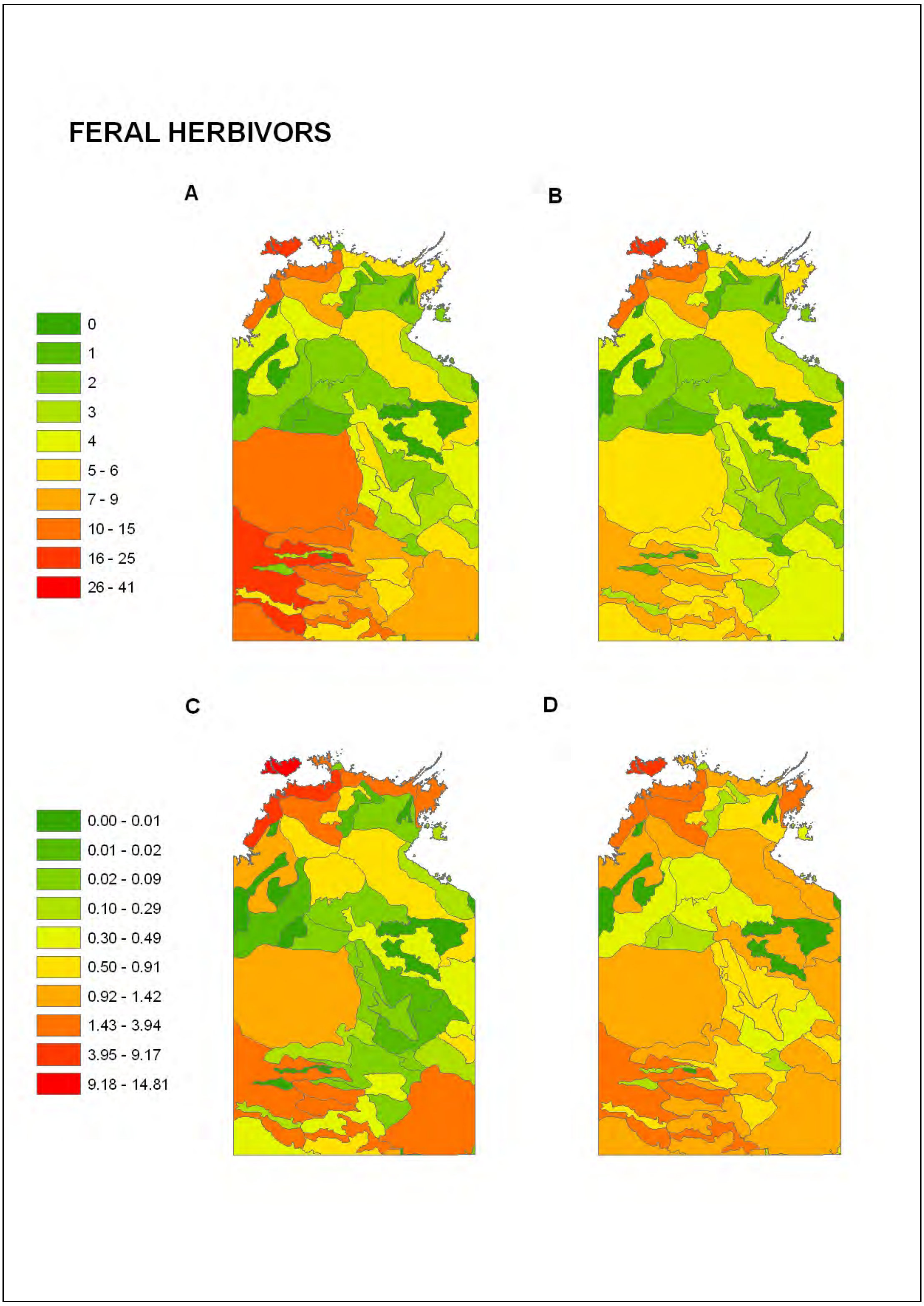

Figure F1.2. Relative impacts of feral herbivores on NT threatened species.

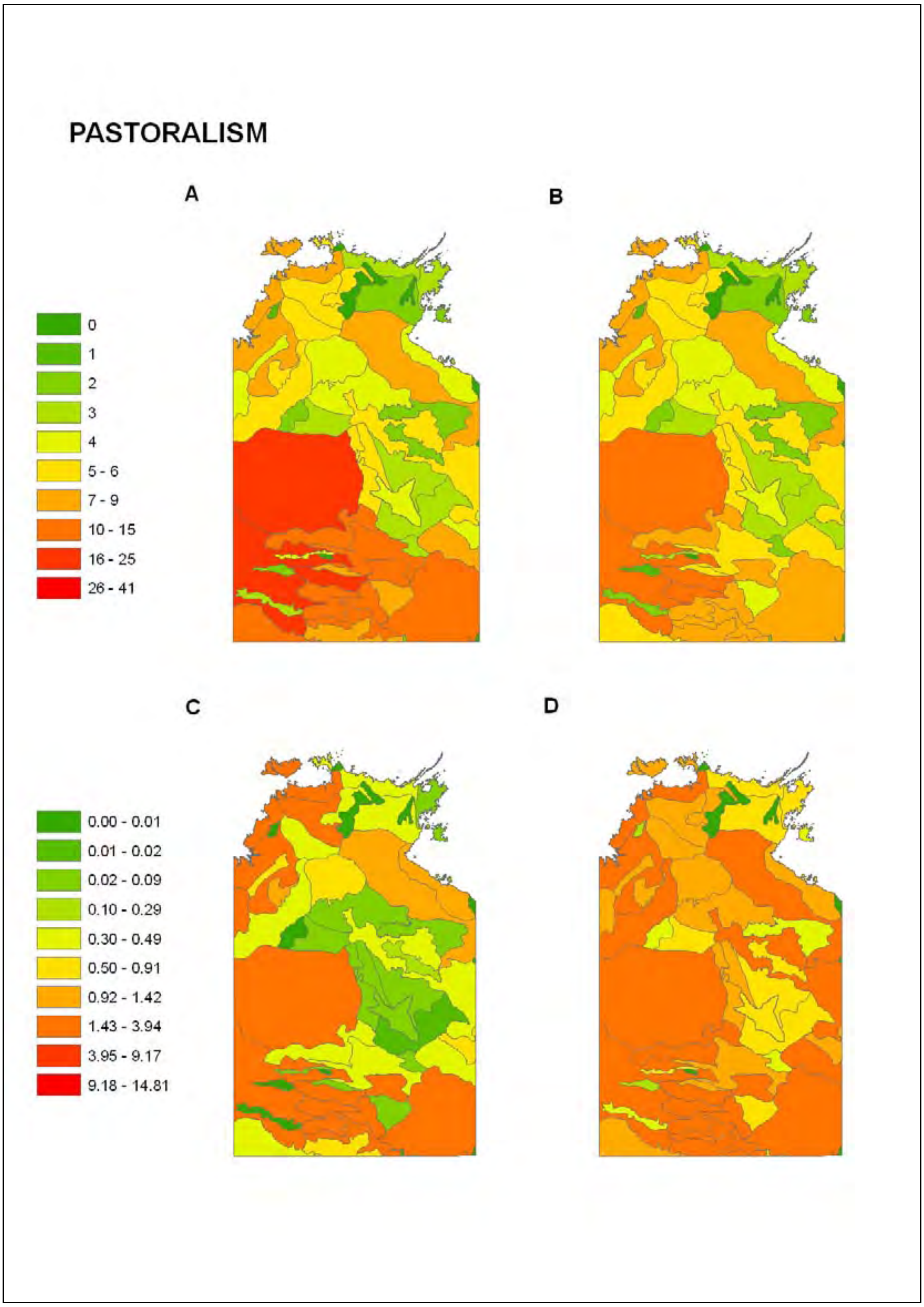

Figure F1.3. Relative impacts of pastoralism on NT threatened species.

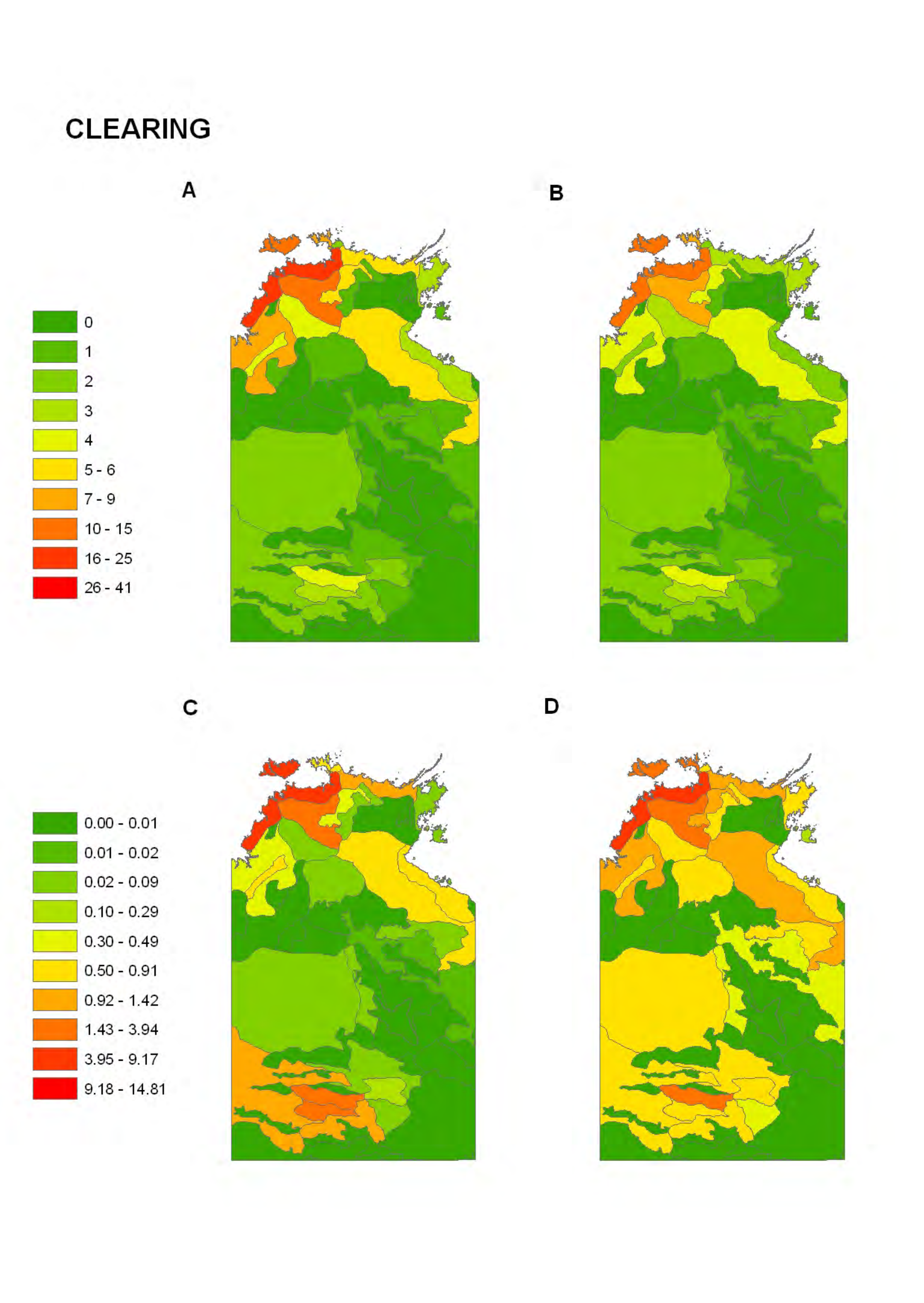

Figure F1.4. Relative impacts of land clearing on NT threatened species.

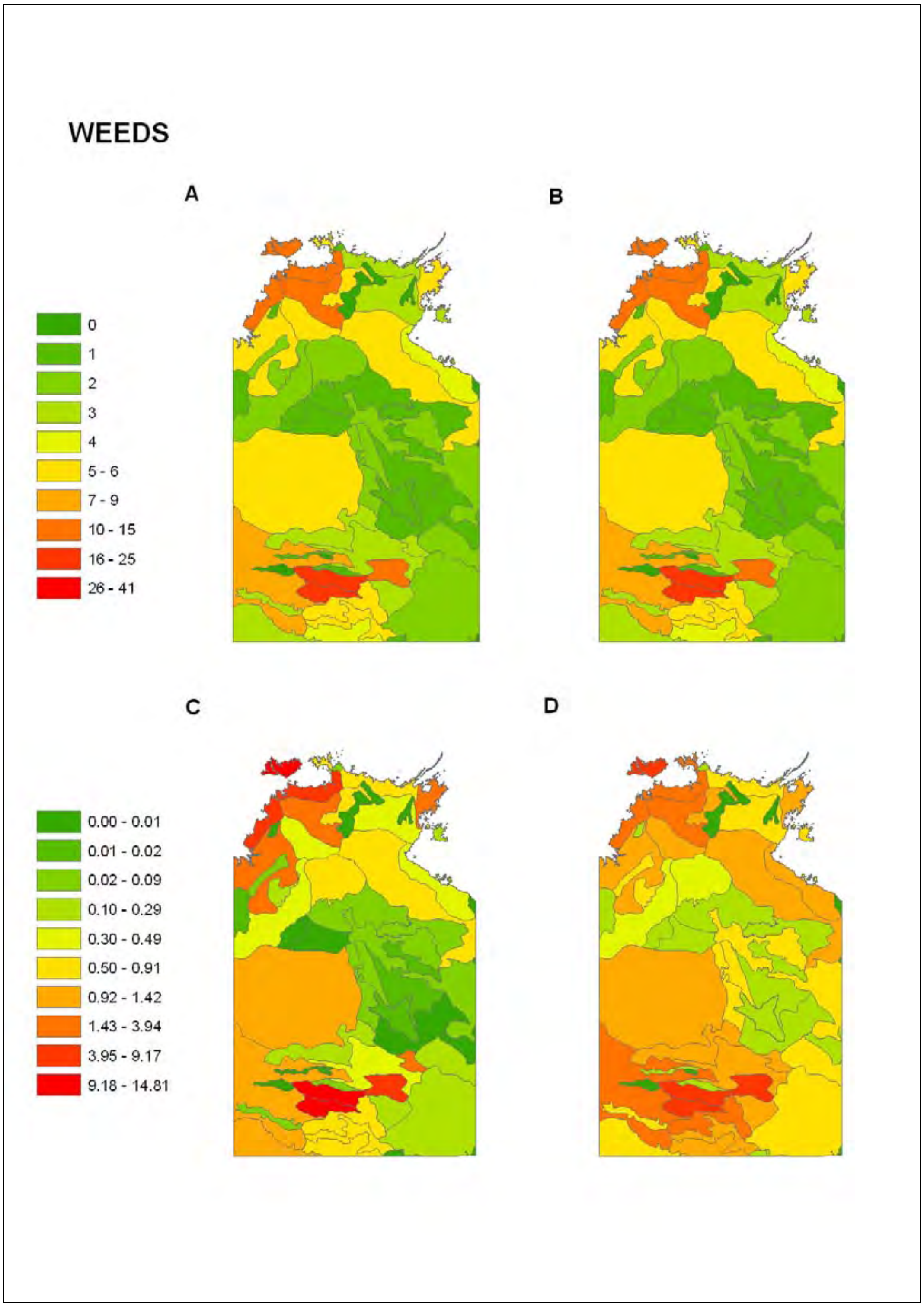

Figure F1.5. Relative impacts of weeds on NT threatened species.

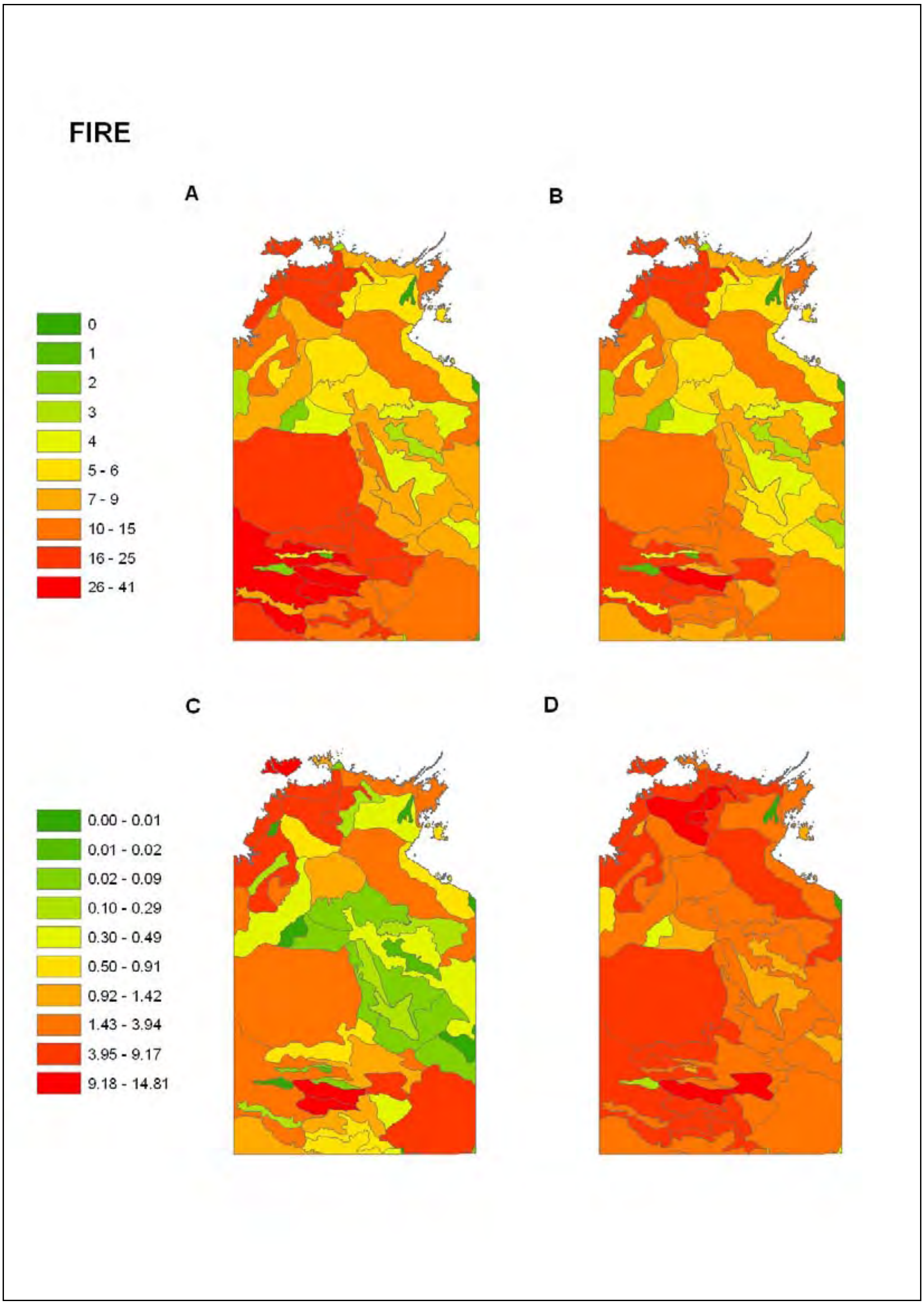

Figure F1.6. Relative impacts of fire on NT threatened species.

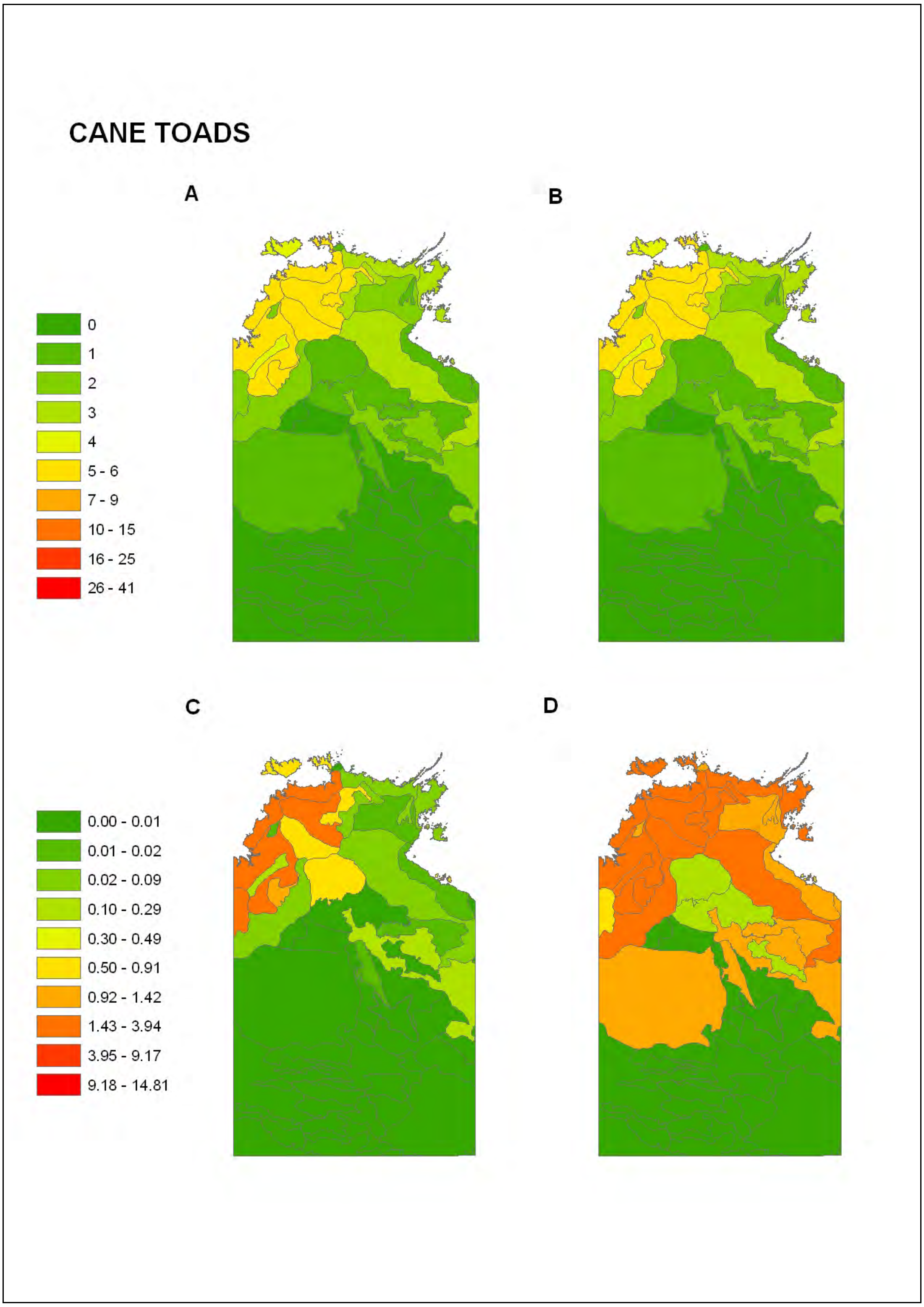

Figure F1.7. Relative impacts of cane toads on NT threatened species.

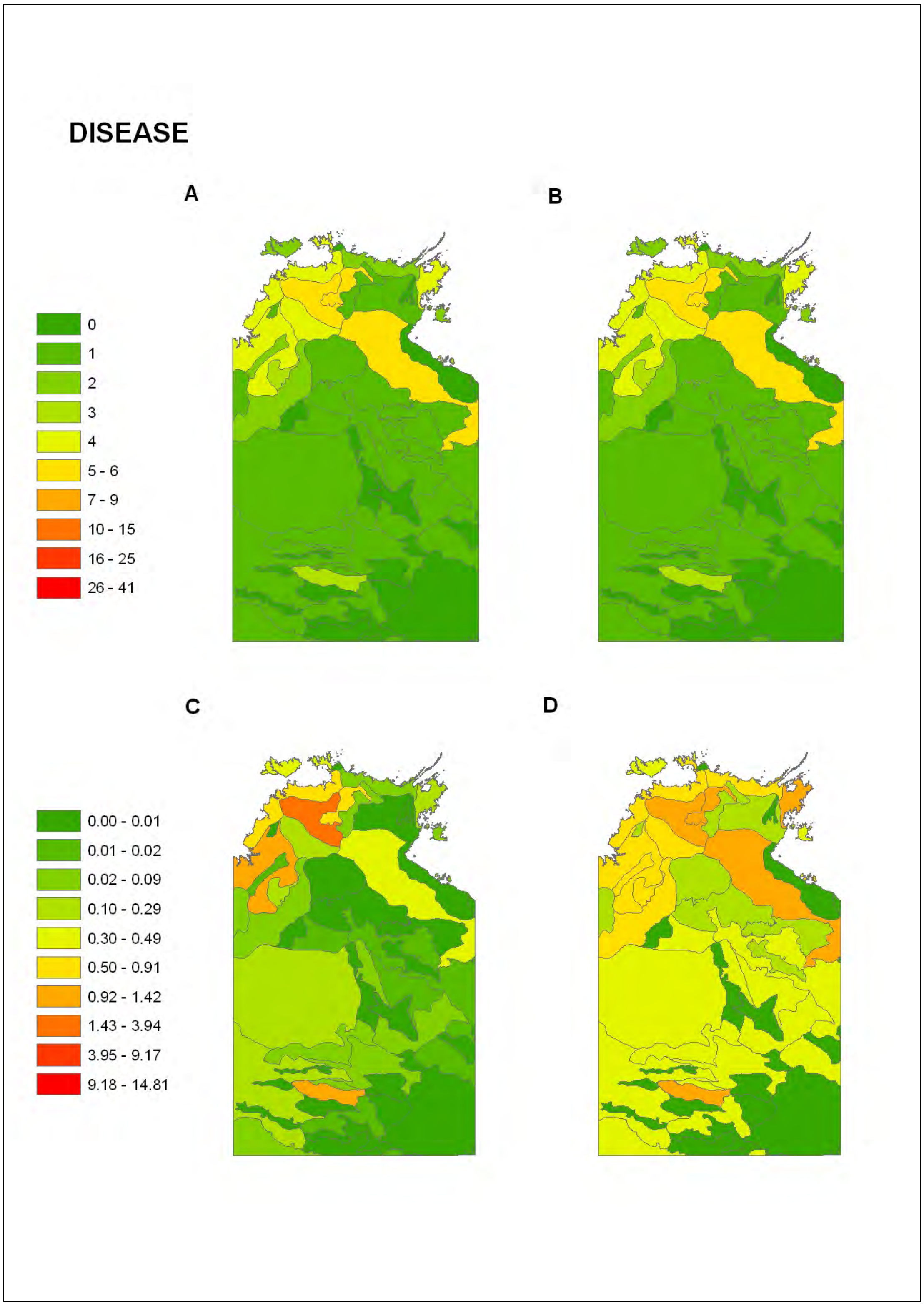

Figure F1.8. Relative impacts of disease on NT threatened species.

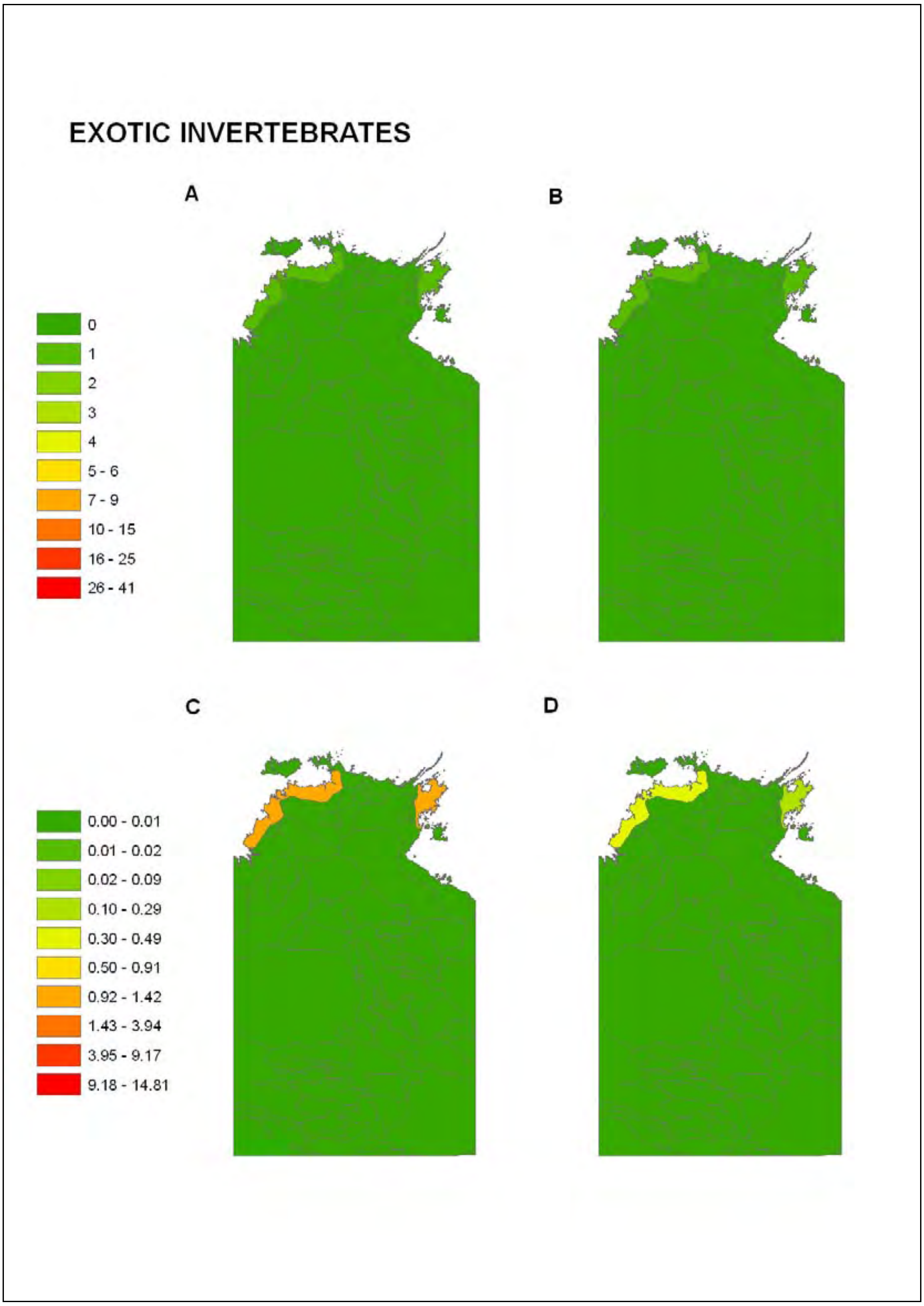

Figure F1.9. Relative impacts of exotic invertebrates on NT threatened species.

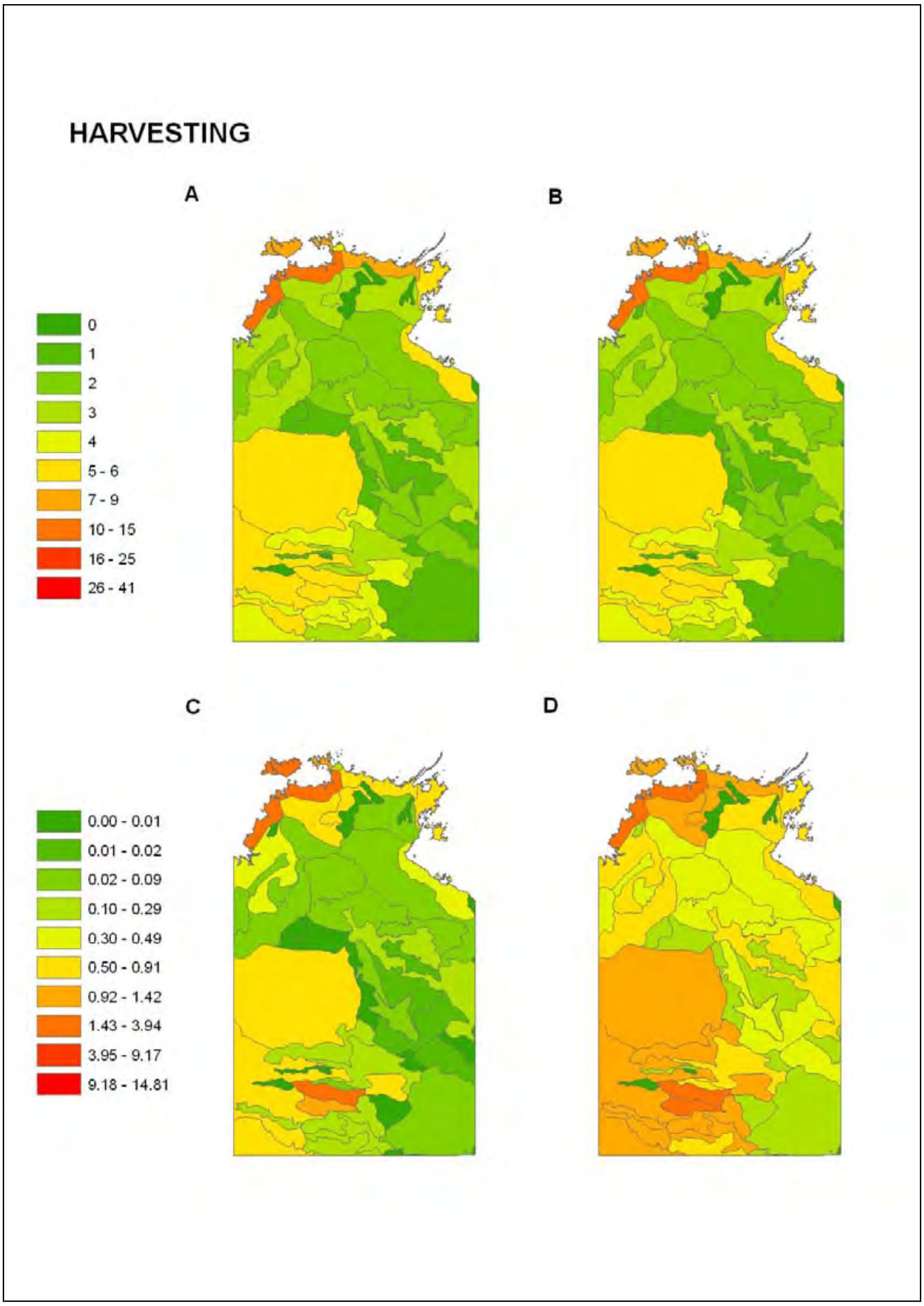

Figure F1.10. Relative impacts of harvesting on NT threatened species.

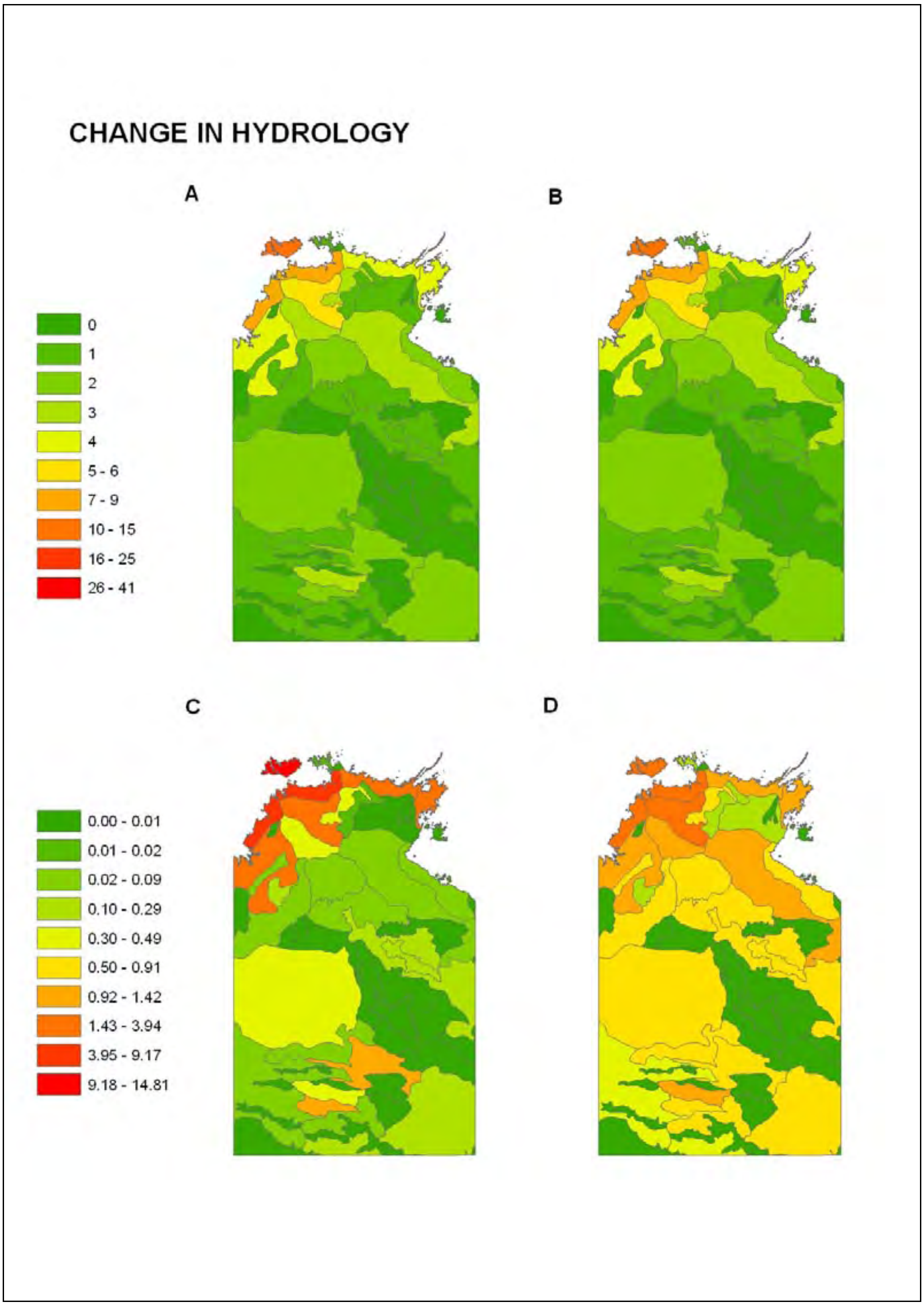

Figure F1.11. Relative impacts of changed hydrology on NT threatened species.

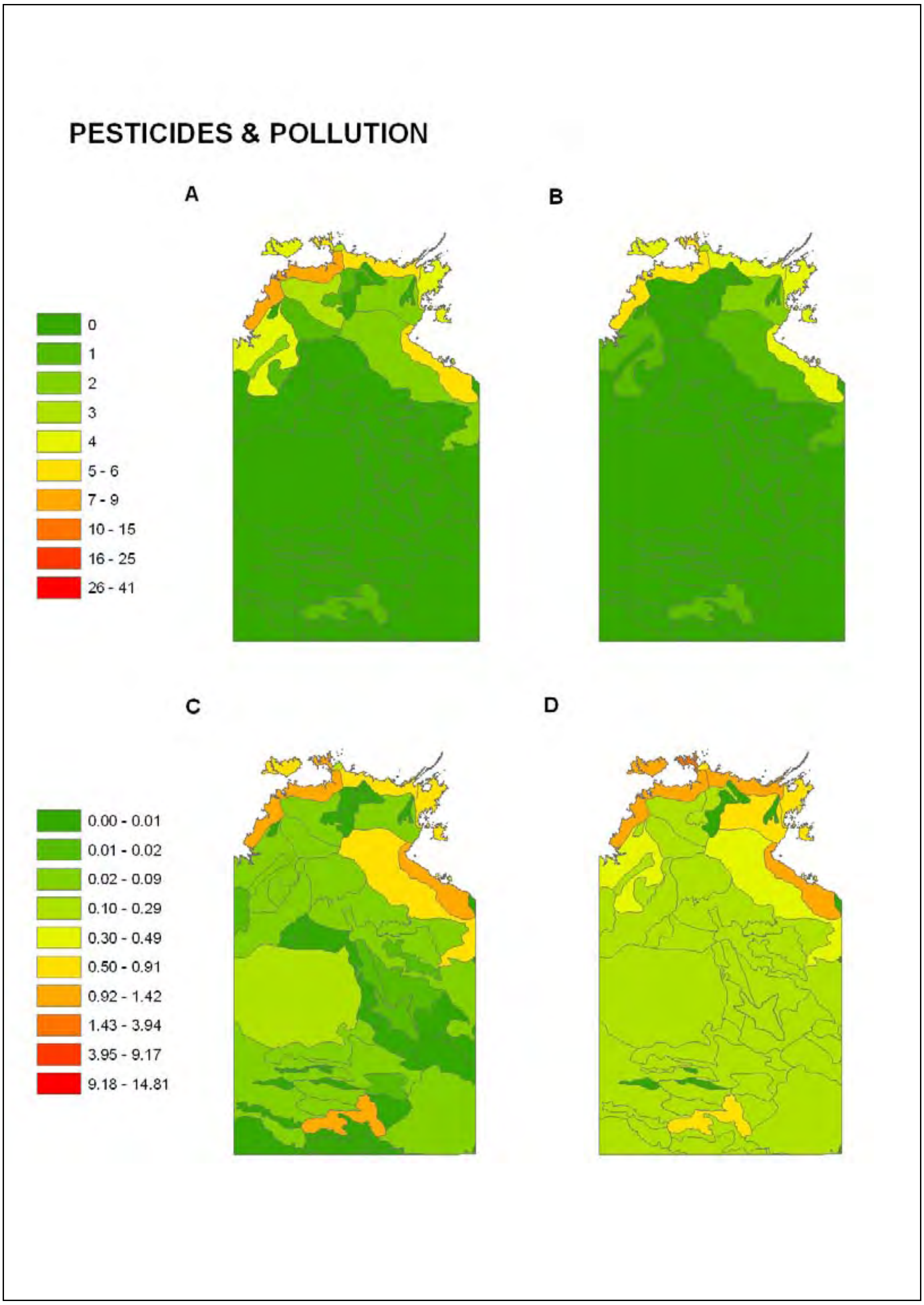

Figure F1.12. Relative impacts of pesticides and pollution on NT threatened species.

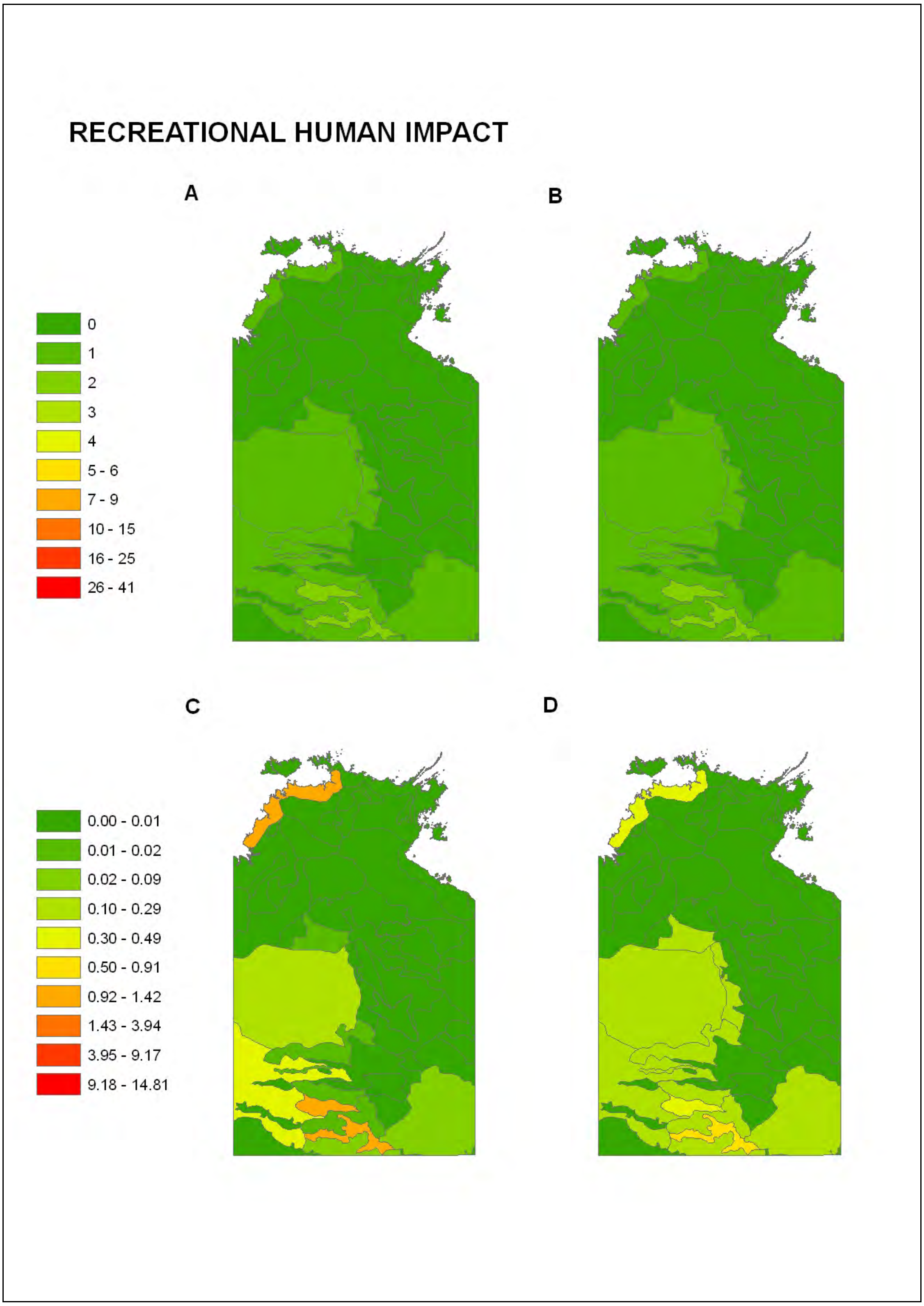

Figure F1.13. Relative impacts of recreational activities on NT threatened species.

Appendix G: Key information gaps – identification and pathways to filling.

Throughout this report, our assessments and analyses have been informed by the most reliable available information. But lack of such information has constrained some assessments of impact, and of management responses. Limitations in knowledge are hardly surprising given the extent of the Territory, the diversity of species and environments, the multiplicity of factors affecting biodiversity, the novelty or relative recency of some threats and management options, the limited amount of biodiversity monitoring undertaken, and the limited previous documentation of management efficacy and costs. With recognition of these factors, we consider that our assessments here should be regarded as pioneering estimates, designed to be tested and refined. But notwithstanding such cautions, our assessments in the body of the table and some appendices, do provide some clear guidelines for prioritisation of management actions and these are likely to be robust and broadly unchanged with advances in knowledge.

In this section, we consider key information gaps, and identify actions that may most effectively remedy such shortcomings. For some factors, these issues are considered in more detail in other sections of the report, notably where quantitative sensitivity analyses were undertaken.

We recognise that there are other information gaps, and other approaches to filling these gaps, but addressing the items on the list below would result in a major advance in our ability to prioritise investments in threat management, and in deriving the most substantial biodiversity benefit from that investment. In compiling here the list of responses to addressing these critical gaps, we can provide no commitment to the resource allocation of NT NRETAS or any other agency. These are recommendations focusing on issues relating to this project, and should be viewed from that context alone.

Issue 1. The relative impacts of threats to at least some components of biodiversity are not well known.

In some cases we have provided expert opinion or subjective assessments of the relative impacts of different threats to NT biodiversity. But, for some cases, it is not yet clear which threats are having the greatest impacts. For example, there is now compelling evidence for the decline of many small to medium-sized mammals in the Top End, but the cause(s) of this decline, or the relative contribution of different threatening factors, remains contestable (Woinarski et al. 2001). Without more clarity concerning the factors that are driving this decline, it is difficult to prioritise remedial management response. The problem is further complicated in that different threats may be acting in a compounded way – for example, predation by cats may be more severe in extensively burnt areas.

Pathway to filling.
Detailed research into the ecology of individual native species, and/or experimentation involving manipulation of putative threatening processes, is the most authoritative mechanism available for assessing the relative impacts of different threats. There are insufficient resources to do this for all NT species, or even for all threatened NT species, so choice of subject species is important. Priority should be given to the most threatened species, species

undergoing the most rapid decline, and/or species that may be most representative of a range of other species.

Issue 2. The impacts of some novel threats are not well known.

Some threats have been operating more or less consistently in the Territory for many decades, such that we now have reasonable information on their characteristics and impacts. But other threats, such as climate change, are operating far more recently and we do not yet have substantial information on their impacts. Even for some established threats, current impacts may not provide a reliable guide to the impacts of a possible intensification of the threats or change in biodiversity response over time. Thus, pastoralism operating well under safe carrying capacity may have relatively few detrimental impacts upon biodiversity; but if herd sizes increase substantially under pastoral intensification, the biodiversity impacts may become much more pronounced. The extent of clearing is currently relatively limited in the NT, but should this be increased substantially, it may exceed some threshold that would then cause major fragmentation responses that fall outside a simple linear extrapolation of existing impacts. The immediate impacts of cane toads on native predators may be devastating (as has been seen for example for the northern quoll over the last decade of the toad's advance through the Top End); but there is some evidence that, over time, some of those predators may be selected, or learn, to avoid toads, or discover ways of eating them that avoid their toxins. If so, these native predator species may eventually return to their previous level of abundance, and the threat of toads will substantially diminish.

Pathway to filling.
There is no simple and satisfactory response to this information shortcoming. But there are a range of useful approaches: (i) seek general trends and rules from places where these threats are already operating; (ii) establish monitoring programs that provide some early warning of the impacts of new threats, or new levels of threats; (iii) where possible, use retrospective analyses (e.g. of responses to previous unusually hot periods) or spatial analyses (e.g. attempt to find sites that currently have similar climates to that projected for our area of interest); (iv) where possible and appropriate, undertake experimental manipulation of threats including to levels that may exceed the current "normal" range, and record biodiversity responses. To some extent, the "Pigeon Hole" pastoral study provided an example of the latter approach, through recording biodiversity responses to a range of grazing intensities, including levels substantially above current "normal" rates (www.mla.com.au/TopicHierarchy/IndustryPrograms/NorthernBeef/The+Pigeon+Hole+Project.htm).

Issue 3. There is little information for some current presumed threats (notably disease).

The incidence of disease and its impact upon NT biodiversity is poorly known, more so than for any other of the threats considered in this project.

Pathway to filling.
There is a range of responses that may contribute to filling this gap. Baseline assessment of the disease status of a wide range of NT biodiversity is required to provide an assessment of what is "normal". The disease status of non-native plants and animals occurring in the NT should also be assessed, and results used to assess the risks of transmission of novel diseases to native plants and animals.

Risk-analysis should be used to identify diseases that may have the most concerning combination of severe impact and likelihood of occurrence, and a management response developed accordingly that considers surveillance, monitoring and the appropriate mix of remedial management actions.

Issue 4. The form of the threat/management response is not well known.

The impacts of some threats operate largely linearly – the more there is of the threat, the greater the impact. But other threats are more complex. Some factors are recognised to pose threats to NT biodiversity, but the response of individual native plant or animal species to the threat or its management may be markedly non-linear, and may contrast between different species. For example, many plant species show responses to fire regimes, but the preferred regime may not be simply frequent burning or fire exclusion, but rather (for example) fire at intervals of 5-10 years, with more or less frequent fires being disadvantageous. The preferred fire regimes are not well known for most NT species.

Further, co-existing species may have contrasting responses and preferred regimes. Balancing the needs of different components of biodiversity creates complex requirements for fire management, that are not well catered for in the relatively simple fire management modeling provided in this report.

Pathway to filling.
The particular responses of individual species to complex threats (such as fire) should be more substantially researched. There is a need to develop more complex decision-support tools for fire (and comparable other threats) management, that can best cater for different and sometimes contradictory needs of multiple species.

Issue 5. We may be unaware of some biodiversity declines, that may be caused by threats not considered here

It is possible that some species in the Territory are declining without us being aware of this decline, and because of causes that we have not considered in this report. Unrecognised decline is quite likely, given that only a small minority of NT native plant and animal species are being monitored.

Pathway to filling.
The most effective means for detecting declines generally is through “ambient” monitoring, that is, a broad-brush approach that is not targeted to particular species already presumed to be of changing status. Such monitoring may also provide early warning of responses to climate change, and insight into other factors that may be affecting biodiversity.

Issue 6. “Safe” levels of threat are poorly known (thresholds and limits).

Some of the analyses considered here evaluated financial costs of controlling threats to a presumed “safe” level, rather than total eradication. However, there is little evidence for what such a safe level may be for particular threats and particular native plant of animal species. This may be especially so for the responses of biodiversity to water extraction or to vegetation clearing. For these threats, regulation typically attempts to impose a threshold or limit –

simplistically, within which the impact is considered acceptable and beyond which the impact is considered unacceptable. Such thresholds are not yet based in the NT on robust evidence.

Pathway to filling.
Similar to Issue 2 above: (i) seek generalisations from situations elsewhere that represent levels of threat beyond that seen currently in the NT; (ii) undertake experimental manipulations for model (small-scale) systems in which resource use (e.g. clearing, water extraction) exceeds the current limited range; and (iii) regulate cautiously to stay within conservatively safe bounds.

Issue 7. The responses of threats to climate change are poorly known.

Issue 2 above noted that the responses of biodiversity to climate change are poorly known, because such change will be largely beyond the current range operating. But climate change will not only affect NT biodiversity directly, it will also affect (and, in many cases, compound) existing threats to biodiversity. The form of this compound impact is poorly known. One likely response is that fire impacts may be more pronounced, given projected increases in temperature.

Pathway to filling.
There is no simple mechanism for filling this gap. A risk analysis of the likelihood of increased incidence or intensity of threats under different climate change scenarios may provide some bounds for possible changes in the existing threats or the likelihood of needed changes in their management prioritisation. Retrospective analysis (e.g. of fire impacts during previous hot periods) may provide some predictive insights.

Issue 8. Social responses to threats are not well known and/or were not well incorporated into our economic models.

Our analyses were largely restricted to assessments of biodiversity impacts of threats, and these assessments contributed to our prioritisation of management responses. But, with the main exception of considering the benefit to pastoralism of control of feral herbivores, we did not provide a substantial consideration of other social factors. But, management resourcing, acceptance and success is likely to be substantially influenced by social factors. For example, in some Indigenous communities, feral animals may now provide a major food resource and proposals for their control or eradication may be unwelcomed. Ideally, a much broader range of factors should have been included in our modeling and prioritisation, including for example the greenhouse gas emission value of differing fire regimes, the production costs and benefits of different weed species, the responses of the community to cat control, and the social and economic benefits to remote Indigenous communities of jobs relating to weed, pest and disease surveillance. Such dimensions are beyond the scope of this particular project, but we note that this project has built a platform and process on which such components can be added.

Pathway to filling.
Many social and economic parameters are more tractable than the biodiversity parameters considered in this report. It should be relatively straightforward to evaluate the relevant social and economic components and include them into the type of modeling used here, albeit with the caution that it may make modeling and prioritisation unwieldy.

Issue 9. The range of management options and techniques will change. Some current threats have no established control mechanisms.

Some techniques for controlling threats have changed little: it is still best practice to shoot feral animals (although even here, such a management technique may change depending for example upon the strength of the animal welfare lobby). But other techniques have changed rapidly, and, optimistically, new techniques may be appreciably less expensive and/or more effective than the techniques considered in this report. These may include new chemicals or more effective biological control agents for the management of weeds. In our analyses, we concluded that broad-scale control of cats (or cane toads) in the NT was prohibitively expensive, but such a conclusion may dramatically change if an effective and host-specific disease was developed and introduced.

Pathway to filling.
Where risk-assessment justifies investment, continue to search for novel control mechanisms for threats that are deemed to have serious environmental consequences but no current cost-effective control measures. Recognise that the existing prioritisation based on cost-effectiveness of current management procedures may change continuously as new techniques are developed.

Issue 10. Investment in threat management should be geographically prioritised.

For most analyses in this report, we have treated the NT as largely homogenous, and assumed that threat mitigation should be undertaken equitably across all regions. (The main exception was for our detailed regional-based analysis of prioritisation of feral animal management.) But for maximising biodiversity returns from investments in threat management, it may be most prudent to concentrate that management on sites of highest biodiversity conservation significance, and/or on sites where there is particular concentration of species affected by that particular threat.

Pathway to filling.
To a reasonable extent, this gap has been filled through a concurrent NHT-supported project, the identification of sites of biodiversity conservation significance in the NT. The results from that project should help prioritise geographic locations where threat management may have the most beneficial impacts upon biodiversity conservation.

Issue 11. Surveillance (for threats not yet present) was not considered in economic models.

Much of the economic input to our prioritisation models was based on estimated costs of current actions addressing current threats. But a general rule for the management of weeds and pests (at least) is that it will almost always be most cost-effective to prevent the entry of new problem species. Surveillance aimed at stopping the entry to the NT of new pests and weeds was not generally considered in our modeling, but should be recognised as a high priority activity, likely to have a very large benefit:cost ratio.

Pathway to filling.

The costs. procedures, locations and success of existing and ideal surveillance programs should be collated and reviewed.

Issue 12. For many management actions, there is little information available on costs and efficacy.

We were able to collate substantial information on costs of a range of management actions, and some information on the efficacy of those actions, for this report; but some of these data are threadbare and there is no reliable data for some threat management actions. This is tantamount to investment without feedback or measure of success.

Pathway to filling.
Generically, there needs to be a commitment to evidence-based management of natural resources. Management actions should be undertaken because they are based on previous knowledge that they work effectively; the costs of those management actions should be carefully compiled; and the successes or failures of the component actions should be evaluated with targeted monitoring of both the threat control and the consequent biodiversity benefit.

References

Woinarski JCZ, Milne DJ & Wanganeen G (2001) Changes in mammal populations in relatively intact landscapes of Kakadu National Park, Northern Territory, Australia. *Austral Ecology* **26**, 360-370.